Lady Bird Lake and Lake Austin status and trends.
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Abstract

Lady Bird Lake and Lake Austin are two valued environmental resources within the City of Austin. Recent concerns about increased development and recreational use along the lakes has prompted the City of Austin to consider instituting policies that would balance the use of the lakes with environmental protection. This report provides a review of the environmental data collected at the lakes to date and any inferences that can be made from the data.

Nutrient data from the lakes show a general degrading pattern of water quality moving in the downstream direction within and between the lakes. There is also evidence that several areas within the lakes would be considered as concerns in supporting the designated use by the Texas Commission on Environmental Quality. Furthermore, temporal trends indicate that the occurrence of phytoplankton in the lakes have recently become more frequent and of a greater concentration. Finally, this report aims to examine the relationship between time lagged nutrient data with flow and the growth of phytoplankton.

Introduction
On May 24, 2012, the City of Austin City Council approved Ordinance 20120524-083. This ordinance established the Lake Austin Task Force, which was charged with developing recommendations concerning the health and function of Lake Austin. In particular, the Lake Austin Task Force was directed to consider and make recommendations concerning the protection of the environment within the Lake Austin watershed. A relevant recommendation is that the City of Austin develop a historical data review for Lake Austin, identify gaps in the data collection on the lake, and look into conducting research to determine the causes of blue-green algae blooms on the lake.

Lady Bird Lake and Lake Austin are two run-of-the-river reservoirs located on the Colorado River running through Austin, Texas. Lake Austin lies immediately below Mansfield Dam on Lake Travis, and runs generally southeast for 20.25 miles. It is impounded by Tom Miller Dam which separates Lake Austin from Lady Bird Lake. Ladybird Lake stretches another 5.4 miles generally southeast through downtown Austin impounded by Longhorn Dam. It is the most downstream reservoir in a chain of seven reservoirs on the Colorado River in central Texas commonly referred to as the Highland Lakes. These reservoirs are highly valued recreational resources and provide important aquatic and riparian habitat for organisms not often found in urban areas. In addition, water from Lake Austin is pulled into two of Austin's drinking water treatment plants. Both the Davis and Ullrich Water Treatment Plant (WTP) are located in the lower portion of Lake Austin. Until 2008, Ladybird Lake was also a source of drinking water through the now decommissioned and deconstructed Green WTP.

Both reservoirs are operated as 'run-of-the-river' or 'pass through' reservoirs by the Lower Colorado River Authority (LCRA), meaning they have no capacity for storing water and the water level fluctuations are minimal. During the summer months, the LCRA has typically released water from Lady Bird Lake to flow downstream to agricultural operations located along the Colorado River. Water from the Highland Lake system, specifically Lake Travis, flows into Lake Austin and Lady Bird Lake during these releases to keep the two reservoirs at a constant level. During such time (typically March 15 – October 15) these two systems take on more riverine characteristics. When water is not being released the systems have an increased retention time and take on more lacustrine characteristics. Due to recent years of drought, the LCRA has altered the management of the summer time releases from Lady Bird Lake to reduce the amount of flow that is allowed downstream. This has created a shift in the conditions in which each lake can be considered to have lacustrine characteristics throughout the year.

The City of Austin (COA) has collected physical, chemical, and biological data within Lady Bird Lake and Lake Austin for many years. For some chemical parameters, the COA has records that extend as far back as 1975 in Lady Bird Lake. Previous reports on the environmental condition of Lady Bird Lake have described trends for nutrient concentrations in the water column, sediment chemistry, phytoplankton blooms, and benthic macroinvertebrate population dynamics (COA 1992, 2001, Herrington 2007). The most recent report identified longitudinally increasing concentrations of ammonia, bacteria, sulfate and total suspended solids (TSS), and decreasing clarity (Secchi disk depth) through the reservoir. Temporal trends indicated an improvement in water quality for metals, dissolved solids, ammonia, total Kjeldahl nitrogen (TKN), total nitrogen, orthophosphorus, and phosphorus. However, there was an indication of increased phytoplankton chlorophyll-a concentrations over time (Herrington 2007). Other COA reports have shown increases in phytoplankton counts up to 2010 in Lake Austin (Richter 2010). In addition, dissolved oxygen problems (instantaneous values <3 mg/L) were noted in the lower portion of Lady Bird Lake (Herrington 2007).

Herrington (2007) indicated that there were no temporal trends in PAHs or PCBs in the sediment in Lady Bird Lake; however, recent work by the United States Geological Survey (USGS) has shown a decrease in PAHs within the sediment of Lady Bird Lake (Van Metre and Mahler 2014). Organochlorine pesticides, including DDT, were reported to be decreasing over time as well. Despite contaminant levels potentially toxic to aquatic life in the sediment, Lady Bird Lake macroinvertebrate community metrics indicated that the communities were not substantially impaired and were less impaired during release seasons (Herrington 2007).

This report functions as an update to the environmental conditions present in Lady Bird Lake and Lake Austin. The primary focus of this report is on the conditions within the water column of each reservoir. Sediment and benthic macroinvertebrate data has not been analyzed but may be the subject of future analysis on the lakes. Phytoplankton communities have been analyzed only within Lake Austin because of the rich data set present within this reservoir. Lady Bird Lake does not have as an extensive data set but may be susceptible to similar algal dynamics as Lake Austin given that water from Lake Austin flows through Lady Bird Lake. As a final objective in this report, the Austin Lake Index was analyzed to determine if the index requires modification.

Data Summary

Water quality measurements have been collected on Lady Bird Lake and Lake Austin since the 1970's. The most consistent measurements throughout this time include dissolved oxygen, temperature, pH, specific conductivity, ammonia, nitrate/nitrite, Total Kjeldahl Nitrogen, orthophosphorus, total phosphorus, and total suspended solids. The locations sampled for these water quality measurements are listed in Table 1 below along with time period of collection and the minimum number of measurements taken per water quality parameter. The minimum number of measurements taken usually referred to measurements of total suspended solids or orthophosphorus. The maximum number of measurements was usually numbered over 1000 and pertained to temperature and dissolved oxygen.

Table 1: Sample site numbers and locations.

Sample Site No	Sample Site Location	Data First Collected	Most Recent Data Collected	Minimum number of site visits
1	The Basin	02/03/1975	12/03/2013	733
2	1st St.	02/03/1975	12/03/2013	369
3	Lamar Blvd.	02/03/1975	05/08/2005	489
4	Mopac Expressway	08/15/1991	10/01/2008	238
5	Red Bud Isle	02/03/1975	12/03/2013	558
561	Tom Miller Dam	10/17/1978	2/19/2014	642
562	Bull Creek	10/17/1978	5/9/2005	68
933	Capital of Texas Hwy	4/17/1995	10/12/2005	39
573	Emma Long Park	10/17/1978	2/19/2014	420
559	Selma Hughes Park	10/17/1978	3/16/1998	44
560	Mansfield Dam	10/17/1978	4/7/2014	183

Location maps of the sample sites are shown below with Lady Bird Lake depicted in Figure 1 and Lake Austin depicted in Figure 2.

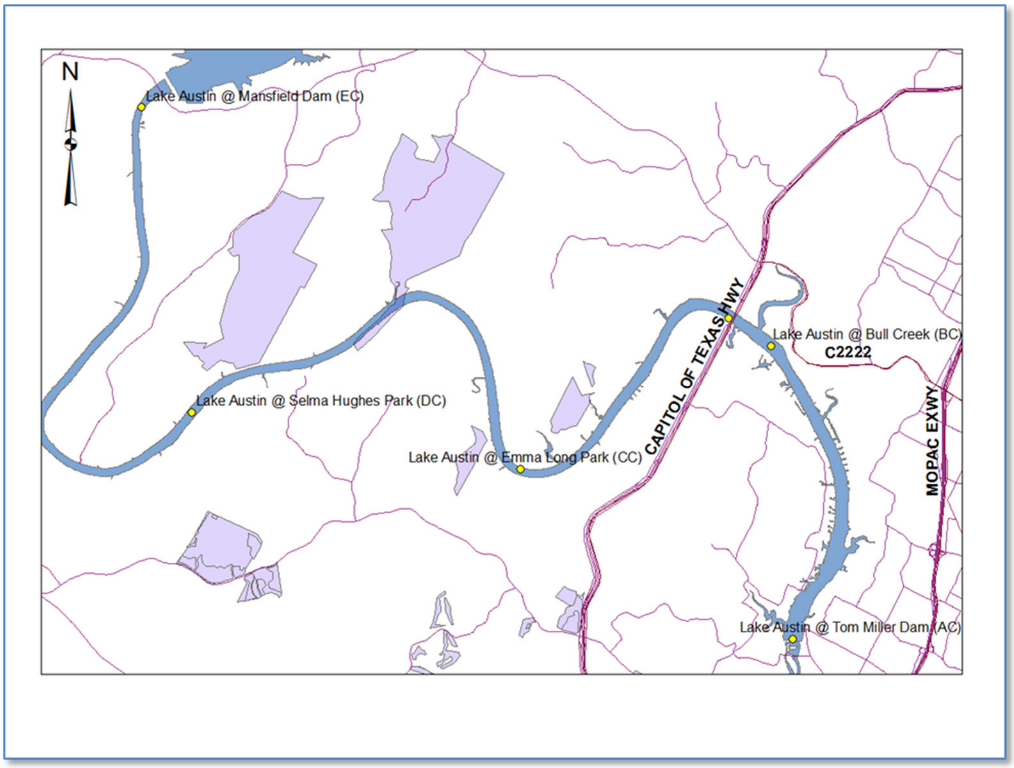


Figure 1 - Lake Austin sample site locations.

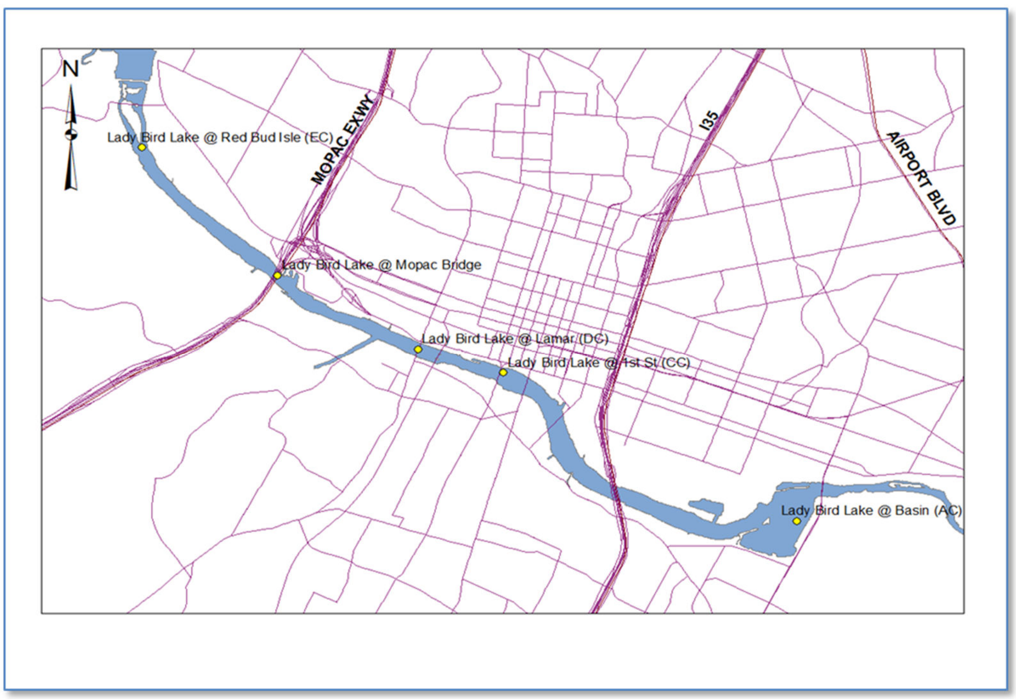


Figure 2 - Lady Bird Lake sample site locations.

Measurements of the physical and chemical nature of the water were taken by several entities and compiled by the City of Austin Environmental Resource Management Section. These entities included United States Geological Survey (USGS), Lower Colorado River Authority (LCRA), Texas Natural Resource Conservation Commission (TNRCC), and the City of Austin Water and Wastewater Utilities. However, the vast majority of the samples were obtained by the City of Austin Environmental Resource Management Section.

In addition to physical and chemical water quality parameters, phytoplankton counts have been collected in Lake Austin from 1992 through 2013 at the Davis Water Treatment Plant (WTP) and the Ullrich WTP which lie between the Lake Austin @ Bull Creek and Lake Austin @ Tom Miller Dam sites. Phytoplankton was either collected from the WTP intake or from the shoreline near each plant. Blue-green algae, green algae, flagellates, diatoms, and total algae were counted in natural units, where colonies and filaments are counted as a single organism, and extrapolated to organisms/mL.

Methods

<u>Water Chemistry Trends</u>

The principal method of analysis of the physical and chemical water quality data was through the construction of confidence intervals of the mean for each parameter at each sample site. Confidence intervals of the mean contain the unknown mean of a sampled population and offer a way to quantify the uncertainty in the estimated mean of the sampled population of each parameter. For this study, the confidence intervals of the mean will serve three purposes. First, the confidence intervals of the mean will be used as comparison between sites. Overlapping confidence intervals indicate similar means between sites and will be used as a rough heuristic for determining whether the estimated means at each of the sites are statistically different. Second, confidence intervals of the mean provide a range by which the average of samples from the same site and parameter will be contained. This is useful in determining whether certain water quality criteria involving averages of the samples are met. Finally, confidence intervals assign a value (or range of values) to each parameter and site. This is crucial for quantifying information for future lake studies *a priori*.

The confidence intervals of the mean are computed via the following equation:

$$[LCI, UCI] = \bar{x} \pm t_{\alpha,N} \cdot s/\sqrt{N}$$

In this equation, LCI and UCI designate the lower and upper confidence interval, respectively; $\bar{x}$ is the sample average, $t_{\alpha,N}$ represents the inverse t-distribution with a probability of α and N degrees of freedom, s is the sample standard deviation, and N is the number of samples taken.

In addition to confidence intervals, lower and upper prediction intervals were computed to estimate the range of the next future measurement. This is in contrast to confidence intervals which construct a range of the characteristic of the sampled population, such as the mean or variance. Because prediction intervals have more uncertainty than confidence intervals, prediction intervals were partitioned into seasons for a more precise prediction. These prediction intervals, which are also useful in quantifying information for future lake studies, can be found in

Appendix A. The equation, which uses the same terms as the previous equation, used to calculate the prediction interval is:

$$[LPI, UPI] = \bar{x} \pm t_{\alpha,N-1} \cdot s / \sqrt{1 + \frac{1}{N}}$$

Since the confidence intervals were used to compare between sites and to compare to TCEQ criteria, which are based on various moving averages, there was no need to partition the confidence intervals into seasons.

Temporal trends in the water chemistry were examined through a linear regression analysis at each sample site for each parameter. This statistical method estimates the mean concentration over time. However, this estimated mean will be highly influenced by a small number of outliers (that is, measurements which are either really high or really low in concentration). Thus a more general trend that focuses strictly on the estimated mean may lead to incorrect estimates of change over time (Terrell et al. 1996, Cade et al. 1999). "Regression quantiles" is a class of statistics developed in the 1970s by two econometricians that is an extension of the linear model for estimating change in all portions of the distribution of a response variable (Koenker and Bassett 1978). This class of statistics has become commonly known as quantile regression and is used to estimate a specified percentile of the distribution of the response variable conditional on a second variable (Cade et al. 1999, Koenker and Machado 1999). Therefore, quantile regression of the water chemistry as the response variable at each site for each parameter was done using the 10^{th} percentile, the 25^{th} percentile, the median, the 75^{th} percentile, and the 90^{th} percentile.

It should be noted that some of the water chemistry data, specifically ammonia, phosphorus, and orthophosphorus, consisted of non-detects (i.e. the concentrations of a parameter were probably less than the Minimum Detection Limit). Although, this typically presents complications in data analysis, these data points were analyzed using a regression on statistics (ROS) routine, which imputed these non-detects with values based on an assumption of a normal or log-normal distribution. This method works for data that are at most 50% non-detects. All of the data sets met this requirement.

<u>Phytoplankton Trends</u>

The biological water quality component examined was primarily the blue-green algae counts within Lake Austin at the Davis and Ullrich WTP because previous reports show that blue-green algae counts have been increasing in Lake Austin over the past decade (Richter 2010). Blue-green algae counts at the Davis WTP and the Ullrich WTP were analyzed for temporal trends with regression analysis and bloom frequency of blue-green algae was examined with logistic regression. Flow and phosphorus concentration impacts on phytoplankton counts (total phytoplankton and blue-green algae) were examined using structural equation modeling, a statistical technique for testing and estimating causal relations.

While the blue-green algae data was not collected at equal time intervals continuously throughout the data set (not collected every day), the data closely resembles a time series data set. There are a number of problems associated with analyzing time series data with ordinary least squares (OLS) linear regression. The most notable problem is the issue of autocorrelation or serial correlation (Kutner et al. 2005). Autocorrelation occurs when model error terms are correlated or strongly related over time. The error terms in an OLS linear regression are assumed to be uncorrelated or independent. A violation of this assumption can lead to the following problems (Kutner et al. 2005):

1. Estimated regression coefficients are unbiased, but have no minimum variance property.
2. Mean squared error may underestimate the variance of the error terms.
3. Standard deviation of the estimated regression coefficient may be underestimated.
4. Confidence intervals and tests using the t and F distributions are no longer strictly applicable.

Due to the problems stated above, the data was explored using a B-spline function of time. Splines are piecewise polynomial functions. When used for regression purposes they do not suffer from the above constraints on the error terms. Applying a spline function typically is a way to smooth data, or extract a trend from noisy data. The amount of smoothness applied to the spline is critical in analysis. Too much smoothing will make temporal dynamics disappear while too little smoothing can lead to spurious conclusions. The B-spline is a commonly used time series smoother, which controls the amount of smoothness by varying the number and location of knots that define the break points between the piecewise polynomials (de Boor 1978). The more knots included in the analysis the less smooth the spline. Regression analysis was done for blue-green algae counts at the Davis and Ullrich WTPs from 1992 through the end of 2013 using a two degree polynomial B-spline of the date with 153 ((number of years x 7) – 1) knots. While the number of knots can be subjective, there is a rule of thumb to allow for seven knots per year of analysis to allow for seasonal changes.

Box-Cox transformation analysis was used for total phytoplankton counts, blue-green algae counts, and phosphate concentrations collected at each WTP as well as flow data collected at Tom Miller dam, downstream of both WTPs. In a Box-Cox transformation analysis, the value of λ is equal to the power that the response variable in a model should be raised to in order to bring the distribution of the response variable closer to a normal distribution (Box and Cox 1964). The 99% confidence intervals for λ included zero for total phytoplankton counts, blue-green algae counts, and flow. When $\lambda = 0$, the most appropriate transformation of the response variable is the log transformation (Kutner et al. 2005). Thus all models related to blue-green algae counts, total phytoplankton counts, or flow at the Davis WTP or the Ullrich WTP on Lake Austin used the log transformations. The phosphate concentrations had $\lambda = -0.05$, which is a difficult transformation to interpret. Because the λ was so close to 0, the log transformation was applied to the phosphate concentrations as well.

More general trends in the blue-green counts over time were also examined through regression analysis at each WTP from 1992 through the end of 2013 using a two degree polynomial B-spline of the date with five knots at the Davis WTP and four knots at the Ullrich WTP. While the seasonal changes will disappear, the long term trends in the data should become much more interpretable as the spline function is much smoother with the decreased number of knots. However, the error variance is not constant over the entire time interval for the blue-green algae data, thus a more general trend that focuses strictly on the mean of the counts may lead to incorrect estimates of change in the counts over time (Terrell et al. 1996, Cade et al. 1999). Quantile regression using blue-green algae as the response variable at each WTP was done using the median, 75th percentile, 80th percentile, and 95th percentile.

In addition to the intensity of blue-green algae blooms, it is important to examine how often algal blooms are occurring on Lake Austin. Thus, trends in the bloom frequency should be examined. One process to compare the bloom frequency or duration from year to year is to count the number of days where the blue-green algae counts are above some threshold value considered to be a bloom threshold. It would be difficult to analyze any statistical difference between the number of days in a bloom in a given year because there are no replications and no variance within years. However, the probability of a blue-green algal bloom occurring on any given day within a year can be used as a surrogate to measure bloom frequency/duration.

Logistic regression can be used to relate binary responses (bloom, non-bloom) to explanatory variables. The outcome from the model is a function that computes the predicted probability of an event occurring based on the value of the explanatory variables. Each day that blue-green algae was collected was marked as either an algae bloom day or a non-bloom day. A day was classified as a bloom day if the blue-green algae count was higher than 300 organisms per mL, a threshold established by the Austin Water Utility. This binary input was placed into logistic regressions for each WTP using the year as an explanatory variable and the predicted probability of a bloom occurring on any given day within a year was found. If the empirical models fit the data well then the predicted probability will increase as the number of days in a year in which a bloom occurs increases.

The models were determined to fit the data with sufficient accuracy if the models were able to accurately predict the occurrence of a bloom on a given day. In other words, the model predicted the occurrence of a bloom when an actual bloom occurred and the model predicted no bloom occurrence when there was no algae bloom. The fit of the models to the data was examined using the *c-statistic* which is the area under a receiver operating characteristic (ROC) curve. The ROC curve plots the fraction of predicted true "events" to total actual "events" (sensitivity) against the fraction of predicted false positive "events" to total actual "non-events" (1 – specificity) (Hastie et al. 2009). The *c-statistic* will range from 0.5 (model randomly predicting the response) to 1.0 (model perfectly discriminating between "events" and "non-events") (McNeil and Hanley 1984).

The predicted probability of a blue-green algae occurring at each WTP in a year was computed from 1992 to 2013 along with the 95% confidence interval for yearly predicted probability. The equation produced from the logistic regression to calculate the predicted probability provides a β for each year. A higher β would relate to a higher predicted probability of algal blooms for a given year. The β was compared between each year to test for differences using the ESTIMATE statement in the PROC LOGISTIC procedure in SAS version 9.2.

A model was created to examine how different water quality parameters may be impacting the phytoplankton counts in Lake Austin at the Davis WTP and the Ullrich WTP. The Austin Water Utility has collected various forms of nitrogen and phosphorus data in conjunction with the collection of phytoplankton counts at each WTP over the years but has collected only phosphate data with any type of regularity. The LCRA has monitored release from Tom Miller Dam for many years. The daily release was used as the daily flow through Lake Austin in the analysis. A multiple regression was considered to model the phytoplankton counts using flow and phosphate concentrations as explanatory variables. However, the flow through (or residence time in) Lake Austin may impact phosphate concentrations in addition to impacting the phytoplankton counts. A form of analysis which allows us to examine the covariance between these observed variables and determine if a theoretical model explains the covariance is known as structural equation modeling, covariance structure analysis, or covariance structure modeling (Kline 2011).

The theoretical model that was examined is best represented by Figure 3 below. Flow was modeled with the flow from the previous day ($\text{Flow}_{T\text{-}1}$) as an explanatory variable. Phosphate was modeled with the flow and the previous day phytoplankton counts ($\text{Phytoplankton}_{T\text{-}1}$) as explanatory variables. Phytoplankton was modeled with flow, phosphate, and previous day phytoplankton counts as explanatory variables. This was done for total phytoplankton counts and blue-green algae counts at each water treatment plant.

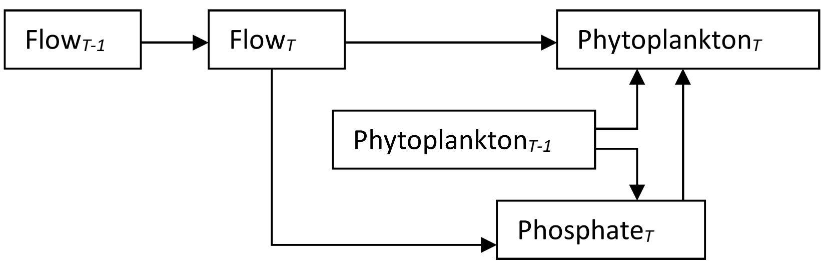


Figure 3: Flow chart for theoretical model used in the structural equation modeling analysis for phytoplankton counts. Subscripts of *T* represent measurements made on a certain day while subscripts of *T-1* represent measurements made one day previously.

Daily phytoplankton counts and phosphate concentrations were needed to perform this analysis, although daily data was not consistently obtained. Imputation is a technique used to fill in missing gaps of data. One imputation technique is to fill in data gaps based on other variables in the data set which are not absent in a particular sample. If the imputation is done only once, the imputed data cannot reflect the uncertainty of the predictions of the missing data (Rubin 1987). Thus multiple imputation using Markov Chain Monte Carlo techniques to impute all missing data points was performed on the data set which included flow, phytoplankton counts, and phosphate concentrations. Multiple imputation replaces the missing data point with a plausible value based on other observed variables many times (Rubin 1976, 1987). The end result is multiple complete data sets. Analysis can be run on the separate data sets and analysis results can be combined to make statistical inferences about the data set.

The number of imputed data sets generated will affect the imputation standard errors and statistical power (Rubin 1987, Graham et al. 2007) which lead to better inferences with increasing imputed data sets. An infinite number of imputations will lead to the lowest standard errors and highest statistical power, but often it is practical to perform only a few imputations. Imputed standard errors are often not appreciably larger than their hypothetical minimum value given a small number of imputations (Enders 2010). The magnitude of a multiple imputation standard error relative to the theoretical minimum is expressed as the relative efficiency (RE):

$$RE = (1 + FMI/m)^{-1}$$

The m is the number of imputed data sets and FMI is the fraction of missing information (Rubin 1987). A relative efficiency close to one indicates that the imputation standard errors are close to their hypothetical minimums. While the relative efficiency comes close to 1.0 fairly quickly, the statistical power rises more slowly. However, optimum statistical power may be reached by as few as 20 imputations if the FMI is not very high (FMI < 0.5) (Graham et al. 2007). The FMI for phosphate was close to 0.90 in the COA data set. To reach a relative efficiency of 0.99 and try to increase statistical power, 90 imputed data sets were created for the analysis.

The structural equation model was examined for each of the 90 imputed data sets. Parameter estimates and standard errors along with model fit statistics from the 90 model outputs were combined to make statistical inferences (Rubin 1987). Fit statistics used to declare that the model fit the data well included a chi-square test of whether the covariance matrix implied by the model is close to the sample covariance matrix, goodness of fit index (GFI), root mean square error of approximation (RMSEA), and the Bentler-Bonnett normed fit index (NFI) (Bentler and Bonnett 1980, Jöreskog & Sörbom 1982, Steiger 1990, Kline 2010). Values close to 1.0 represent good fit for the GFI and the NFI, while values close to 0.0 indicate good fit for the RMSEA.

Austin Lake Index Analysis

The City of Austin Watershed Protection Department (COA WPD) has recently developed an Austin Lake Index (ALI) that scores Lady Bird Lake and Lake Austin based on metrics of water quality, sediment quality, physical habitat, benthic macroinvertebrates, eutrophication, and aquatic vegetation. Richter (2011) describes sampling and index calculation methods for the ALI. The index scores each component on a score of 0 (worst) to 100 (best) and is meant to compare environmental conditions on each reservoir over time. Results from each sub-index and the overall ALI for each lake in each year since the program's inception is presented.

The Lower Colorado River Authority (LCRA) currently collects phytoplankton samples on Lake Austin four times a year at Mansfield Dam, Emma Long Park, and Tom Miller Dam. Samples are collected in April, June, August, and October. While there is not much flexibility in the current sampling schema, samples are collected in warmer months where phytoplankton blooms are thought to be more likely if nutrients are available. Chlorophyll *a*, blue-green algae counts, green algae counts, diatom counts, and chrysophyte counts from these data are input into the ALI (Richter 2011) to calculate the change in eutrophication on Lake Austin from year to year.

When making comparisons from year to year, it is important to analyze samples that were collected in similar conditions. For example, it would not be wise to compare one sample collected during a bloom to a sample collected the next year while not experiencing a bloom and conclude that the lake is becoming less eutrophic. Collecting a higher number of samples should increase the probability that the data is being collected in similar conditions; however, sampling is costly both in time and money. This report examined how consistent our current sampling scheme is with regards to the Eutrophication sub-index in the ALI and if there is a change in protocol that could lead to more consistent and representative Eutrophication sub-index.

First, phytoplankton data from the Davis WTP was used to examine which months have the highest probability of bloom occurrence for total phytoplankton, blue-green algae, and diatoms. This was done by taking the mean and standard deviation from each algal group and declaring any phytoplankton count that was one standard deviation higher than the mean count as a bloom. A logistic regression was performed on the data using the month and year as explanatory variables. The fit of the logistic models was examined using the *c-statistic* which is the area under a ROC curve. The probability of a bloom occurrence was examined for April, June, August, and October to determine in which currently collected month is the probability of an algal bloom the highest. Lake Austin was represented in this analysis instead of Lady Bird Lake due to the existence of almost daily phytoplankton community data collected at the WTPs. The Ullrich WTP was not analyzed as the analysis would most likely be redundant.

Currently one phytoplankton grab sample from April, June, August, and October is collected for input to the ALI (ALI). If data was collected every day in April, June, August, and October, the ALI score would fully represent the phytoplankton conditions present in the lake during these months. Sampling each day is cost and time prohibited, but increasing the frequency of sampling within April, June, August, and October might allow the COA WPD to capture phytoplankton communities in more than one state (bloom vs non-bloom) if they existed and produce more accurate ALI scores. Phytoplankton data from the Davis WTP in 2012 was used to calculate the Blue-green sub-index, Diatom sub-index, and Eutrophication Index for each day where data was available as described in the ALI methodology (Richter 2011). The year 2012 was chosen because it was a recent year with more variability in the predicted bloom probability than 2013. A data scheme was set up where days were randomly combined into 1, 2, 3, 4, and 5 samples a month for April, June, August, and October to determine if an increased sampling frequency would reduce the variability possible in sub-indices and index scores.

In addition to sampling more frequently in months where COA WPD already samples, sampling in months other than April, June, August, or October might increase the ability of the ALI score to represent phytoplankton biomass and community composition over a given year. In order to determine how additional sampling in months other than April, June, August, or October would affect the ALI scores, one hundred random samples were generated from data collected at the Davis WTP for the months of April, May, June, July, August, September, October, and November. These months were chosen because they have higher predicted probabilities in some years including 2012 which was the year used for analysis. A Blue-green sub-index, Diatom sub-index, and Eutrophication sub-index were calculated for each random sample. The base score for each random sample was calculated as the average score from April, June, August, and October. Each monthly score was averaged into the base score until all combinations of additional monthly sampling were analyzed.

As a final analysis, the ALI was also examined over the period from 1992 through 2013 using phytoplankton data at the Davis WTP to determine how the index would have changed over this time frame. An index was calculated for every day in April, June, August, and October for each year from 1992 through 2013 based on a combination of blue-green algae and diatoms. A second index was calculated which was strictly based on blue-green algae in order to compare against the current index. Average index scores were calculated each year using indices calculated in April, June, August, and October.

Results

<u>Water Chemistry</u>

The estimated means and variability of water chemistry within each lake are provided in this section. Also, water chemistry results are compared against TCEQ water quality standards and screening criteria as they are used in Clean Water Act 305(b) and 303(d) reports. Finally, temporal trends within lakes and longitudinal trends between and within the lakes will be examined.

Within Lake Variability of the Estimated Mean at Lady Bird Lake

Table 2 below shows 95% confidence intervals of the mean for various parameters at different sites within Lady Bird Lake. Downstream sites are generally more degraded than upstream sites. Statistically significant differences in the means between locations were recognized for all parameters except for orthophosphorus, which remained constant at around 0.04 mg/L and total dissolved solids at approximately 315 mg/L. One interesting result is that the Lamar Blvd. site (#3) shows elevated levels of nitrate/nitrite. This level diminishes by the next downstream site, 1st Street (#2). Similarly, conductivity is high at Lamar Blvd. (#3) and then decreases by the next downstream site.

Table 2: Confidence Intervals of the Mean for chemical and physical parameters in Lady Bird Lake. Sample site numbers run from upstream (5) to downstream (1).

Parameter	**Sample Site No.**				
	5	4	3	2	1
Phosphorus (mg/L)	0.025-0.035	0.033-0.045	0.031-0.038	0.036-0.049	0.038-0.050
OrthoPhosphorus (mg/L)	0.007-0.048	0.025-0.033	0.027-0.037	0.022-0.034	0.023-0.029
Total Kjeldahl Nitrogen (mg/L)	0.29-0.33	0.29-0.34	0.33-0.38	0.33-0.39	0.36-0.41
Ammonia (mg/L)	0.04-0.04	0.04-0.07	0.05-0.07	0.05-0.06	0.06-0.09
Nitrate/Nitrite (mg/L)	0.23-0.27	0.22-0.28	0.35-0.41	0.29-0.33	0.24-0.28
Dissolved Oxygen (mg/L)	7.86-8.10	7.87-8.17	7.49-7.74	7.69-7.91	6.91-7.13
Temperature (^{o}C)	19.68-20.21	19.22-19.94	19.59-20.12	20.09-20.68	20.89-21.34
pH (standard units)	7.74-7.78	7.60-7.67	7.54-7.61	7.69-7.73	7.64-7.67
Conductivity (µS/cm)	522-532	544-560	569-583	515-525	544-555
Total Dissolved Solids (mg/L)	307-332	N/A	310-334	285-325	305-330

Chloride (mg/L)	44-47	32-37	37-42	32-36	41-45
Sulfate (mg/L)	30-31	25-27	30-32	25-28	28-30
***E. coli* (mpn/100mL)**	2-31	2-94	22-473	13-374	4-184

Table 3 summarizes the results using post-hoc tests to indicate whether there was a statistically significant difference in sample sites, which sites showed the biggest difference, and the average difference in the mean between the highest and lowest means.

Table 3: Differences for physical and chemical parameters between sites in Lady Bird Lake.

Parameter	Average Change in Mean Between Sites with Highest and Lowest Mean Conc.	Site Comparison	Statistically Significant
Phosphorus (mg/L)	0.01	1 to 5	Yes
OrthoPhosphorus (mg/L)	0.00	NA	No
Total Kjeldahl Nitrogen (mg/L)	0.06	1 to 5	Yes
Ammonia (mg/L)	0.03	1 to 5	Yes
Nitrate (mg/L)	0.13	3 to 5	Yes
Dissolved Oxygen (mg/L)	1.00	1 to 5	Yes
Temperature (°C)	1.10	1 to 5	Yes
pH (standard units)	0.20	3 to 5	Yes
Conductivity (µS/cm)	50	3 to 5	Yes
Total Dissolved Solids (mg/L)	15	NA	No
Chloride (mg/L)	10	2 to 5	Yes
Sulfate (mg/L)	5	4 to 5	Yes

Within Lake Variability of the Estimated Mean at Lake Austin

Tables 4 and 5 below show confidence intervals of the mean for physical and chemical parameters at different sites within Lake Austin and statistically significant differences between Lake Austin sites, respectively. The spatial trends for Lake Austin were more ambiguous than that of Lady Bird Lake. For instance, the estimated means of dissolved oxygen and temperature were lowest at the uppermost portion of the lake at Mansfield Dam (site 560) and highest at the midpoint of the lake at Capital of Texas Highway (Site 933). Statistically significant differences were seen for all parameters except for ammonia, which had a constant mean of around 0.04 mg/L throughout the lake.

Table 4: Confidence Intervals of the Mean for physical and chemical parameters in Lake Austin. Sample site numbers run from upstream (560) to downstream (561).

Parameter	**Sample Site Number**					
	560	559	573	933	562	561
Phosphorus (mg/L)	0.025-0.054	0.012-0.016	0.025-0.04	0.023-0.093	0.021-0.029	0.027-0.056
OrthoPhosphorus (mg/L)	0.011-0.016	0.010-0.013	0.012-0.017	0.017-0.030	0.013-0.017	0.014-0.019
Total Kjeldahl Nitrogen (mg/L)	0.35-0.45	0.33-0.44	0.35-0.44	0.27-0.38	0.37-0.47	0.40-0.48
Ammonia (mg/L)	0.04-0.05	0.02-0.05	0.03-0.06	0.04-0.05	0.03-0.04	0.04-0.05
Nitrate/Nitrite (mg/L)	0.16-0.18	0.14-0.20	0.12-0.15	0.11-0.14	0.17-0.24	0.13-0.16
Dissolved Oxygen (mg/L)	6.79-7.35	6.58-7.37	8.00-8.21	7.95-8.48	7.53-7.88	7.33-7.54
Temperature (^{o}C)	16.65-17.31	17.24-18.86	18.41-18.91	19.91-21.05	18.66-19.63	18.57-19.01
pH (standard units)	7.87-7.92	7.89-7.99	7.96-7.99	8.01-8.06	7.94-7.99	7.88-7.90
Conductivity (µS/cm)	496-510	542-584	512-523	458-469	554-578	523-532
Total Dissolved Solids (mg/L)	256-280	347-413	286-308	226-250	301-359	303-320
Chloride (mg/L)	50-54	N/A	49-52	41-46	25-41	52-57
Sulfate (mg/L)	30-33	N/A	32-35	29-31	21-41	35-39
***E. coli* (mpn/100mL)**	1-31	N/A	2-17	2-13	5-412	1-14

Table 5 summarizes the results using post-hoc tests to indicate whether there was a statistically significant difference in sample sites, which sites showed the biggest difference, and the average difference in the mean between the highest and lowest means.

Table 5: Differences for physical and chemical parameters between sites in Lake Austin.

Parameter	Average Change in Mean Between Sites with Highest and Lowest Conc.	Site Comparison	Statistically Significant
Phosphorus (mg/L)	0.05	559 to 933	Yes
OrthoPhosphorus (mg/L)	0.01	559 to 933	Yes
Total Kjeldahl Nitrogen (mg/L)	0.1	933 to 561	Yes
Ammonia (mg/L)	0.00	NA	No
Nitrate (mg/L)	0.08	933 to 562	Yes
Dissolved Oxygen (mg/L)	1.1	559 to 933	Yes
Temperature (^oC)	3.5	560 to 933	Yes
pH (standard units)	0.15	560 to 933	Yes
Conductivity (µS/cm)	100	933 to 562	Yes
Total Dissolved Solids (mg/L)	140	933 to 559	Yes
Chloride (mg/L)	22	562 to 561	Yes
Sulfate (mg/L)	7	933 to 561	Yes

Within Lake Variability by Depth

Significant differences of samples taken at different depths of the lake were also evaluated. These different depth measurements were taken on the same day, allowing paired differences between depths. Results from this paired difference analyses showed that dissolved oxygen was consistently about 1 mg/L higher at the surface of the lake than at depth. Similarly, temperature measurements were about 1 to 2^{o}C higher at the surface of the lake than at depth. The pH, conductivity, and the nutrients showed very little differences with respect to depth.

Water Quality Criteria Comparisons

The condition of Lady Bird Lake can also be assessed by comparing lake concentrations to water quality criteria in 30 Texas Administrative Code §307 along with TCEQ guidance for water quality assessment (TCEQ 2012). Under this guidance, assessments of stream segments or reservoirs fall under one of two methods: *use attainment* or *concern assessment*. These methods assume a certain probability, p, of the number of samples exceeding a certain threshold for each water quality parameter out of the total number of samples gathered. If the actual number of samples exceeding the threshold is larger than that predicted with probability, p, with an error rate of 20%, then that stream segment or reservoir is moved to an appropriate compliance category by the TCEQ. The binomial distribution is used to calculate the number of samples exceeding the threshold predicted by using p.

Lady Bird Lake and Lake Austin are assessed under the *use attainment* criteria for the following parameters: water temperature, pH, dissolved solids, and E. Coli. Under *use attainment* criteria, *p* is assumed to be 10% for placing the stream on the 303(d) list by the TCEQ. To place the stream as merely a concern under the *use attainment* criteria, *p* is assumed to be 8%.

Nutrient concentrations for Lady Bird Lake and Lake Austin were classified under *concern assessment* criteria. This criterion classifies streams as a concern if *p* is calculated to be greater than 20%. Note that streams are not placed on the 303(d) list under *concern assessment* criteria.

The thresholds for *use attainment* and *concern assessment* vary by stream. The thresholds for Lady Bird Lake and Lake Austin for the various parameters are as follows:

- Under *use attainment*, the average of the samples for the past seven years is compared to the following segment specific water quality threshold in 30 TAC §307.10 Appendix A:
  100 mg/L for Chloride (Cl^{-1});
  75 and 100 mg/L for Sulfate (SO_4^{-2}) in Lake Austin and Lady Bird Lake, respectively;
  400 and 500 mg/L for TDS in Lake Austin and Lady Bird Lake, respectively; and
  32.2 ºC (90 ºF) for Temperature.

- Under *use attainment*, the geometric mean of the samples is compared to the 30-day geometric mean criteria for primary contact recreation 1 in 30 TAC §307.7(b)(1)(A)(i).:
  126 colonies/100mL for E.Coli

- Dissolved oxygen criteria is also provided in 30 TAC §307.10 Appendix A as Segment 1429 (LadyBird) is designated as having Exceptional Aquatic Life Use. This level of designated use has a dissolved oxygen criteria of not less than 5.0 mg/L on a mean basis and 4.0 on a single sample basis over any 24-hour averaging period (30 TAC §307.7(b)(3) Table 3). The *use attainment* criterion is used for this parameter.

- Similarly to the other parameters for which there are segment specific standards the assessment of high and low pH is done by the *use attainment* criterion using the range of 6.5 to 9.0 SU as listed in 30 TAC §307.10 Appendix A in comparison to the median pH in the mixed layer for each sample event.

- Under *concern assessment*, the average of the samples is compared to the following segment specific nutrient concentration thresholds under Table 3.10 (Screening Levels for Nutrient Parameters) of the 2012 Guidance for Assessing and Reporting Surface Water Quality (TCEQ 2012):
  0.37 mg/L for Nitrate;
  0.11 mg/L for Ammonia-Nitrogen;
  0.66 mg/L for Total Phosphorus; and
  7.56 µg/L and 5.00 µg/L for Chlorophyll-a in Lady Bird Lake and Lake Austin, respectively, under 30 TAC §307.10 Appendix F.

The confidence intervals developed in the previous section can be used as a proxy for determining whether the averages of the samples are below the first two categories in the *use attainment* criteria above. That is, confidence intervals of the mean below the criteria would indicate a good probability that the average of the samples for the past seven years would also be below the criteria. For example, the confidence interval for chloride was between 41 and 45 mg/L in the Basin (Sample Site No. 1). This is below the criteria of 75 mg/L and would thus not have likely exceeded this threshold for more than with a probability, *p*, of 10%.

Table 6: Use attainment criteria exceeded for Lady Bird Lake.

Sample Site No	Chloride	Sulfate	TDS	Temp.	pH	DO	*E. coli*
1	No	No	No	No	No	No	Yes
2	No	No	No	No	No	No	Yes
3	No	No	No	No	No	No	Yes
4	No	No	-	No	No	No	No
5	No	No	No	No	No	No	No

Based on the confidence intervals of the mean provided in Table 2, it appears that the average of the samples for the past seven years has been below the criteria for Chloride, Sulfate, TDS, and Temperature. Similarly, dissolved oxygen and pH maintain fairly constant means within the thresholds limits. *E. coli* would probably have been a concern for Lady Bird Lake downstream of Lamar due to the high variability in its counts (Table 6).

Table 7 below shows the ratio of the number of times the criteria in the *concern assessment* category above was exceeded to the total number of samples collected.

Table 7: Estimated probability, *p*, of exceedances of concern assessment criteria for Lady Bird Lake.

Sample Site No	Nitrite-Nitrogen	Ammonia-Nitrogen	Total Phosphorus	Chlorophyll-a
1	20%	13%	0.2%	44%
2	29%	7%	0.1%	28%
3	40%	8%	0.00%	18%
4	18%	7%	0.00%	16%
5	19%	3%	0.1%	19%

The results, which are shown in the Table 8 below, indicate that nitrite-nitrogen would cause Lady Bird Lake at downstream sites to be classified as a concern, and chlorophyll-a would also cause Lady Bird Lake at the Basin (#1) and at 1st Street (#2) to be classified as a concern.

Table 8: Concern assessment criteria excceded for Lady Bird Lake.

Sample Site No	Nitrite-Nitrogen	Ammonia-Nitrogen	Total Phosphorus	Chlorophyll-a
1	No	No	No	Yes
2	Yes	No	No	Yes
3	Yes	No	No	No
4	No	No	No	No
5	No	No	No	No

The condition of Lake Austin can also be assessed in the same manner as Lady Bird Lake. Based on the confidence intervals of the mean provided in Table 4, it appears that the average of the samples for the past seven years has been below the criteria for chloride, sulfate, and temperature. Similarly, pH and dissolved oxygen show means that are consistently within the threshold limits. One possible exception is dissolved oxygen at Mansfield Dam (Site #560), which shows a mean of 4.78 mg/L during the winter.

Table 9: Use attainment criteria exceeded for Lake Austin.

Sample Site No	Chloride	Sulfate	TDS	Temp	pH	DO	*E. coli*
561	No	No	No	No	No	No	No
562	No	No	No	No	No	No	Yes
933	No	No	No	No	No	No	No
573	No	No	No	No	No	No	No
559	-	-	Yes	No	No	No	-
560	No	No	No	No	No	Yes	No

For concern assessment, Lake Austin would not exceed the criteria (Table 11).

Table 10: Estimated probability, *p*, of exceedances of concern assessment criteria for Lake Austin.

Sample Site No	Nitrite-Nitrogen	Ammonia-Nitrogen	Total Phosphorus	Chlorophyll-a
561	7%	6%	0.8%	22%
562	14%	3%	0%	-
933	1%	2%	0%	17%
573	4%	5%	0.4%	14%
559	8%	4%	0%	-
560	3%	4%	0.7%	4%

Table 11: Concern assessment criteria exceeded for Lake Austin.

Sample Site No	Nitrite-Nitrogen	Ammonia-Nitrogen	Total Phosphorus	Chlorophyll-a
561	No	No	No	No
562	No	No	No	-
933	No	No	No	No
573	No	No	No	No
559	No	No	No	-
560	No	No	No	No

Spatial Variability of the Estimated Means between Lady Bird Lake and Lake Austin

In addition to the confidence interval comparisons, a graphical analysis was used to look at spatial differences in water chemistry parameter means. The first step in exploratory data analysis is usually to graph the raw data. Spatial patterns in the levels of the various measurements between the lakes were examined by looking at Figures 4 through 8. Each graph shows the concentration levels proceeding downstream from left to right. Figure 6 shows the mean phosphorus and orthophosphorus concentrations throughout Lake Austin and Lady Bird Lake. The mean concentrations of orthophosphorus appear to be increasing from an average mean of 0.01 mg/L at the upstream station of Lake Austin to about 0.03 mg/L at the downstream station of Lady Bird Lake. The mean concentrations of phosphorus are a little more inconsistent with the highest mean concentration at Loop 360 of 0.06 mg/L. The detection limit for orthophosphorus and phosphorus is 0.02 mg/L.

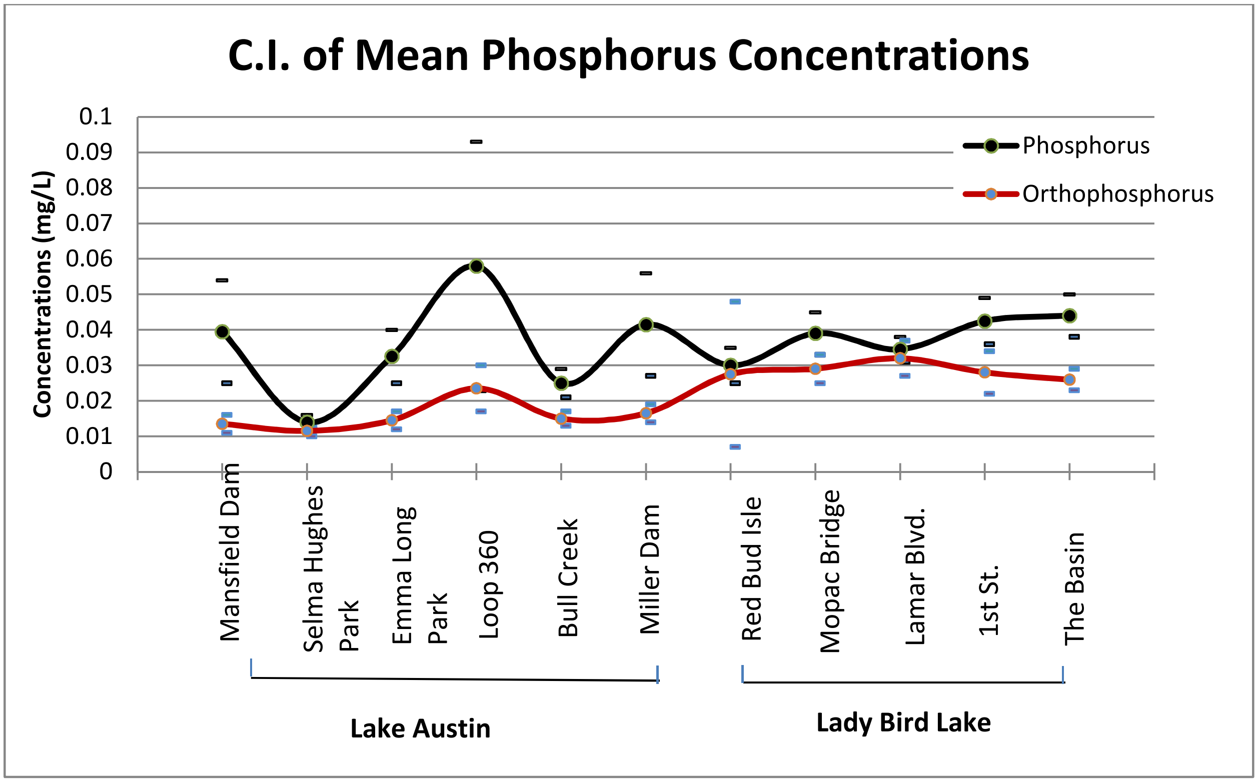


Figure 6: Mean concentrations with confidence intervals for total phosphorus and orthophosphorus concentrations at sites in Lady Bird Lake and Lake Austin.

In the next figure (Figure 7), mean ammonia concentrations remain fairly constant throughout both lakes at around 0.07 mg/L with very small variances. Mean concentrations of nitrate/nitrite are almost twice as high in Lady Bird Lake than Lake Austin with a peak mean concentration at Lamar Boulevard of 0.40 mg/L. The mean concentrations of Total Kjeldahl Nitrogen are also fairly constant between the two lakes at around 0.40 mg/L; however, mean concentrations upstream of the Tom Miller Dam are around 1.5 times higher than the downstream of the dam.

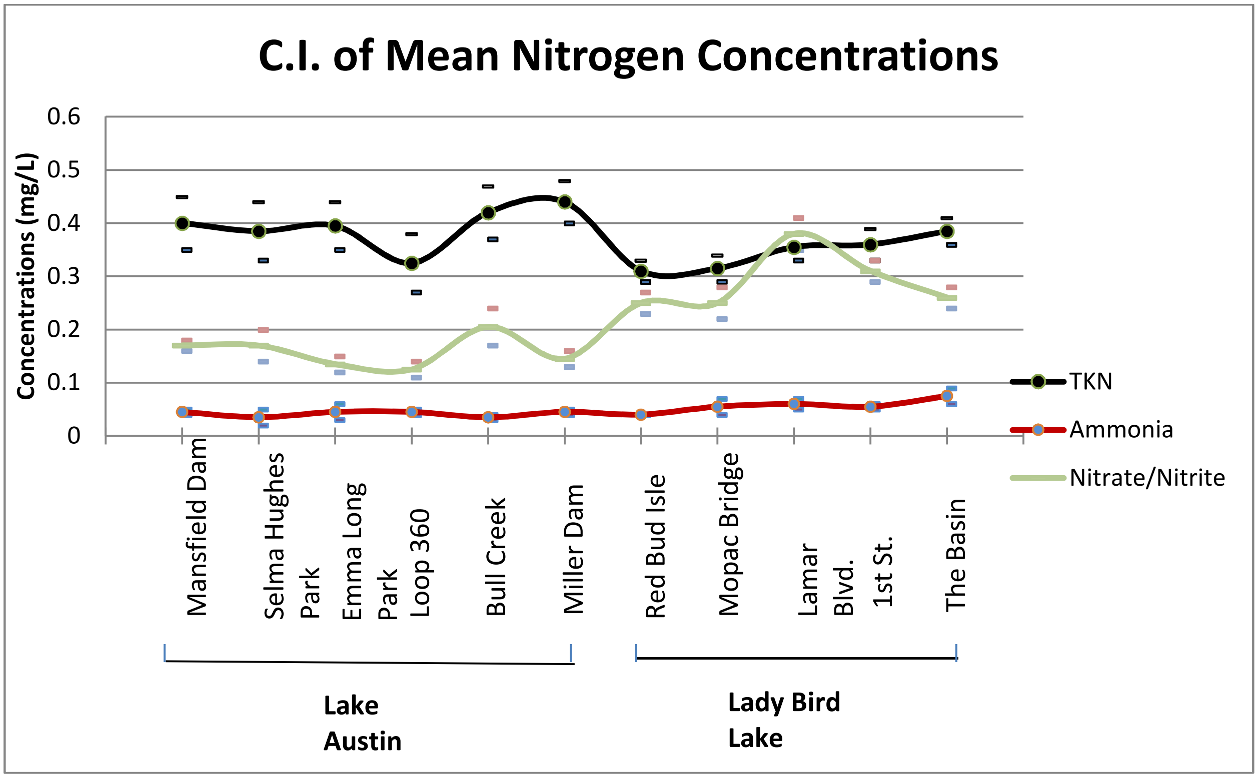


Figure 7: Mean concentrations with confidence intervals for nitrate/nitrite, ammonia, and total Kjeldahl nitrogen (TKN) at sites in Lady Bird Lake and Lake Austin.

The spatial trends of temperature and dissolved oxygen were shown together in Figure 8 due to the typically inverse relation between the two. That is, as temperature increases, dissolved oxygen will decrease. However, this does not appear to hold true in Lake Austin. Both temperature and dissolved oxygen follow each other up to about the Mopac Bridge, where these water quality parameters diverge.

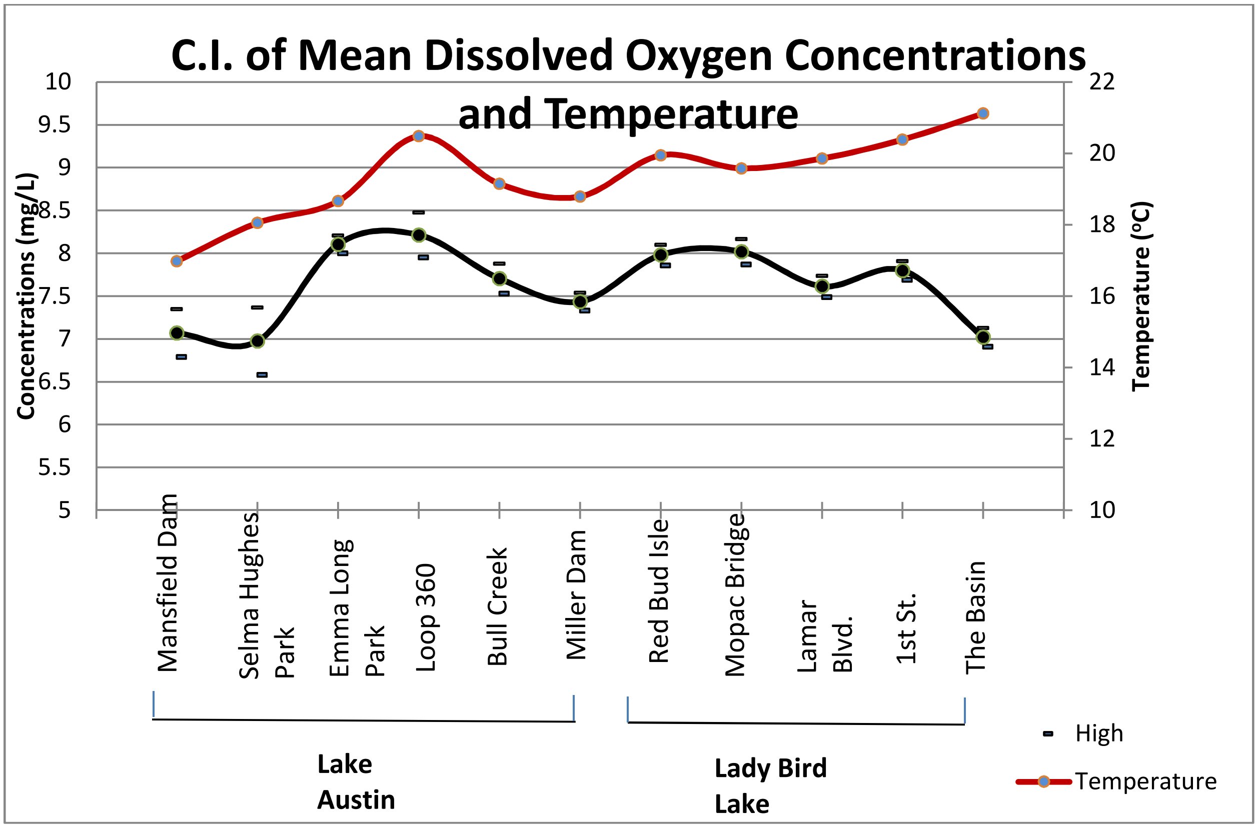


Figure 8: Means with confidence intervals for dissolved oxygen concentrations and instantaneous temperature readings at sites in Lady Bird Lake and Lake Austin.

The spatial trends for pH and conductivity were graphed together, but do not appear to show any correlation between the two (Figure 9). The mean pH levels at the most upstream stations in Lake Austin are approximately at 8.0 standard units and become more neutral at the downstream station in Lady Bird Lake. The mean levels of conductivity do not appear to show any spatial trend. However, at the Loop 360 station, the mean conductivity appears lower than the rest of the stations. This correlates with the high amounts of phosphorus, temperature, or dissolved oxygen.

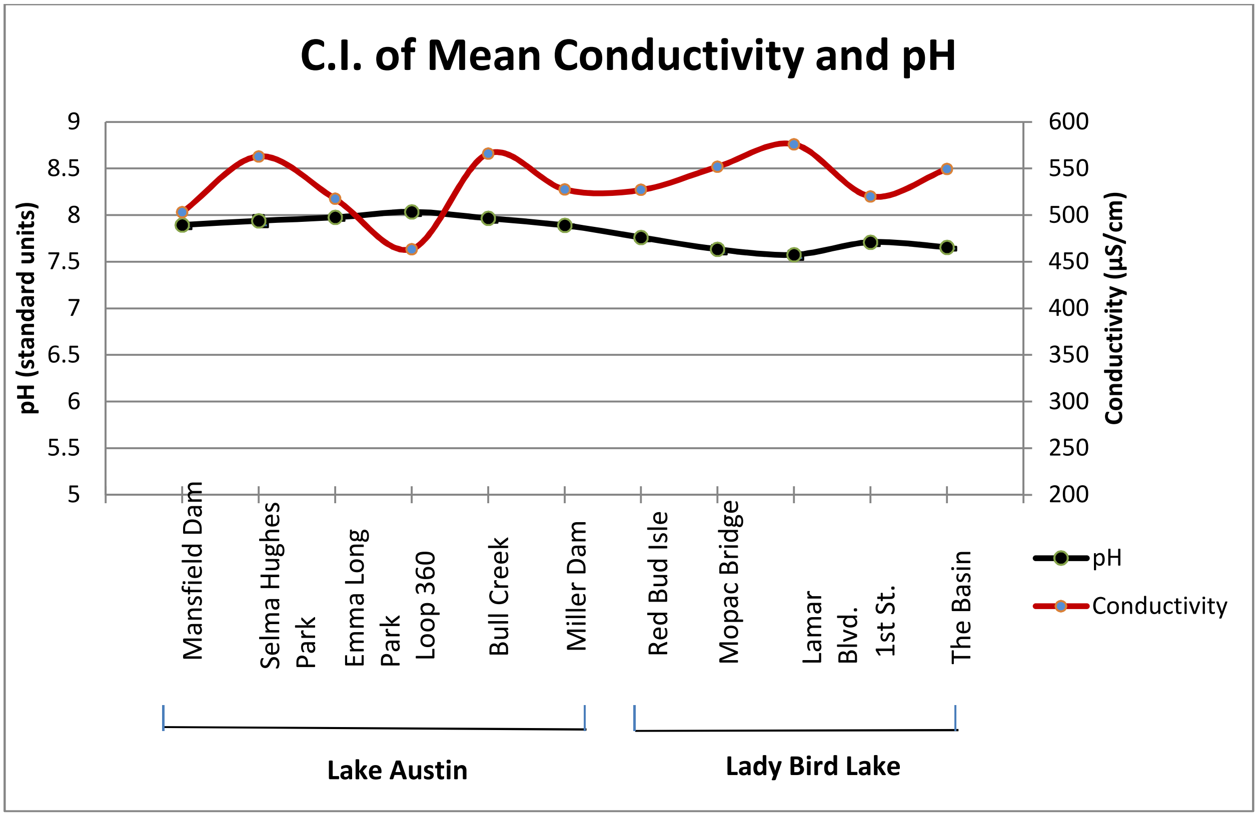


Figure 9: Means with confidence intervals for instantaneous conductivity and pH reading taken at sites in Lady Bird Lake and Lake Austin.

The spatial trends for chloride and sulfate appear to be closely correlated (Figure 10). With the exception of concentrations around Bull Creek, Lake Austin generally has higher concentrations of the analytes than Lady Bird Lake. The highest concentrations of the analytes occur upstream of the Miller Dam.

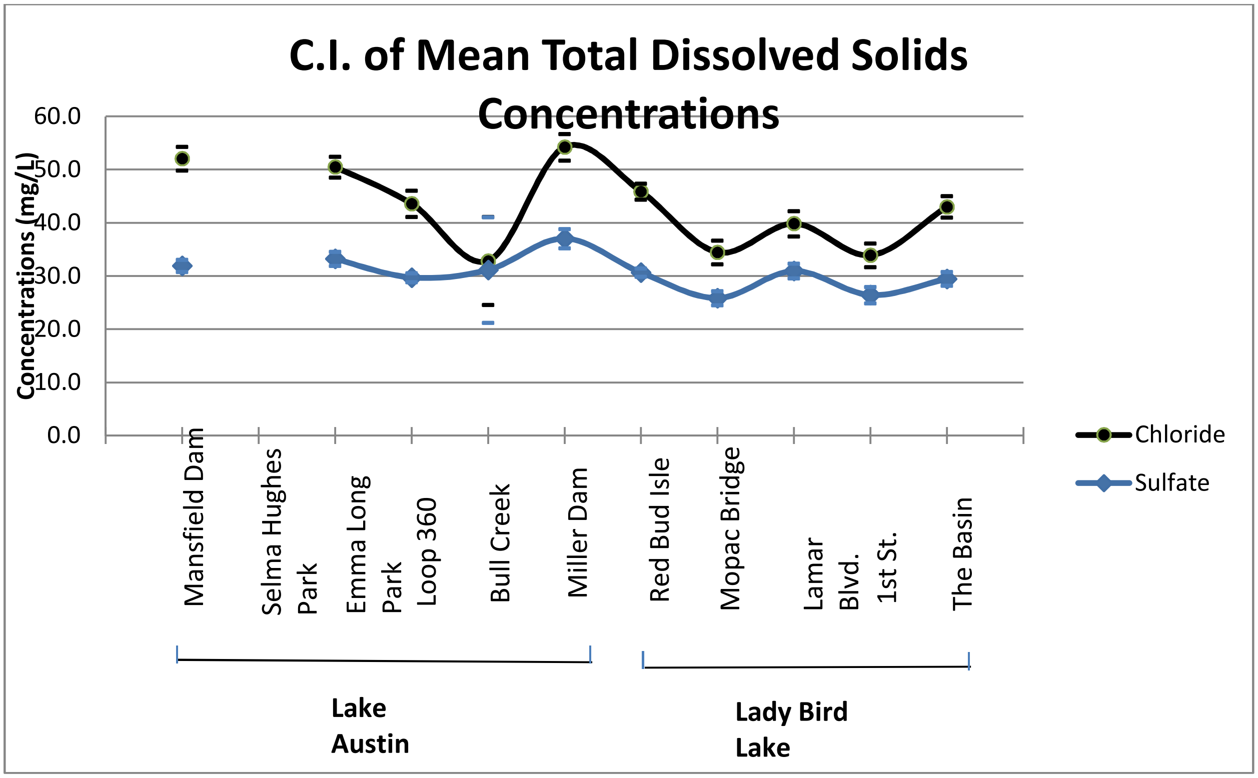


Figure 10: Mean concentrations with confidence intervals for chloride and sulfate at sites in Lady Bird Lake and Lake Austin.

Temporal Trends in Lady Bird Lake and Lake Austin

Trends in water quality concentrations were also analyzed over time (Table 12). Most of the nutrient concentrations in Lady Bird Lake showed either a decreasing or no trend over time. However, temperature and chlorophyll-a concentrations were both increasing at every Lady Bird Lake site while dissolved oxygen and conductivity were both decreasing at every Lady Bird Lake site.

Table 12: Temporal Trends in the Median for physical and chemical parameters in Lady Bird Lake. Sample site numbers run from upstream (5) to downstream (1).

Parameter	**Sample Site No.**				
	5	4	3	2	1
Phosphorus (mg/L)	None	None	None	None	None
OrthoPhosphorus (mg/L)	Decreasing	None	None	None	None
Total Kjeldahl Nitrogen (mg/L)	None	None	None	None	Decreasing
Ammonia (mg/L)	None	None	Decreasing	Decreasing	None
Nitrate/Nitrite (mg/L)	Decreasing	None	None	Decreasing	Decreasing
Dissolved Oxygen (mg/L)	Decreasing	Decreasing	Decreasing	Decreasing	Decreasing
Temperature (°C)	Increasing	Increasing	Increasing	Increasing	Increasing
pH (standard units)	None	None	None	None	None
Conductivity (µS/cm)	Decreasing	Decreasing	Decreasing	Decreasing	Decreasing
Chlorophyll-a	Increasing	Increasing	Increasing	Increasing	Increasing

The temporal trends at Lake Austin are less clear (Table 13). Like Lady Bird Lake, the conductivity appears to be decreasing while chlorophyll-a concentrations are increasing at all Lake Austin sites. However, nutrient trends are mixed. Site 562 (Lake Austin at Bull Creek) has the most number of increasing nutrient parameter (Phosphorus, Orthophosphorus, TKN, and Ammonia). The rest of the sites show either a decreasing or no temporal trend in nutrient concentrations, with the exception of Site 933 (Lake Austin at Loop 360) which shows an increasing trend in phosphorus.

Table 13: Temporal Trends in the Median for physical and chemical parameters in Lake Austin. Sample site numbers run from upstream (560) to downstream (561).

Parameter	Sample Site Number				
	560	573	933	562	561
Phosphorus (mg/L)	Decreasing	Decreasing	Increasing	Increasing	None
OrthoPhosphorus (mg/L)	None	None	None	Increasing	Increasing
Total Kjeldahl Nitrogen (mg/L)	None	None	None	Increasing	Increasing
Ammonia (mg/L)	Decreasing	Decreasing	Decreasing	Increasing	Decreasing
Nitrate/Nitrite (mg/L)	None	Decreasing	None	None	Decreasing
Dissolved Oxygen (mg/L)	Increasing	Increasing	None	None	Increasing
Temperature (ºC)	None	None	None	None	None
pH (standard units)	None	None	None	None	Increasing
Conductivity (µS/cm)	Decreasing	Decreasing	Decreasing	Decreasing	Decreasing
Chlorophyll-a	Increasing	Increasing	-	-	Increasing

Trophic Status of the Lakes

As part of Section 314 of the Clean Water Act, all states are required to classify lakes according to trophic state. This trophic status is determined by regression equations, which produces the Trophic Status Index (TSI); that is, a number between 0 and 100. This scale reflects the continuum of the trophic status of lakes from oligotrophic (TSI less than 35) to mesotrophic (TSI between 35 and 45) to eutrophic (TSI between 45 and 55) to hypereutrophic (TSI greater than 55). Generally, on this scale, lower numbers indicate clearer waters and less biological activity (Carlson 1977).

There are three separate metrics that can be used to determine the TSI. Measurements of chlorophyll-a, total phosphorus, and Secchi disk depth are each calculated along the TSI scale. TSI(Chl-a), TSI(TP) and TSI(SD) are can be used to determine trophic condition, changes over time, efficacy of restoration efforts, and spatial differences within or between reservoirs. The metrics also serve as additional verification or comparison to measurements and trends collected for this study. Combined with predictive water quality models, future TSI under various scenarios of water quality controls can also be estimated. Figures 4 and 5 show the TSI metrics, as measured biennially by the Texas Commission on Environmental Quality, for Lady Bird Lake and Lake Austin.

Figure 4: Trophic Status Index for Lady Bird Lake

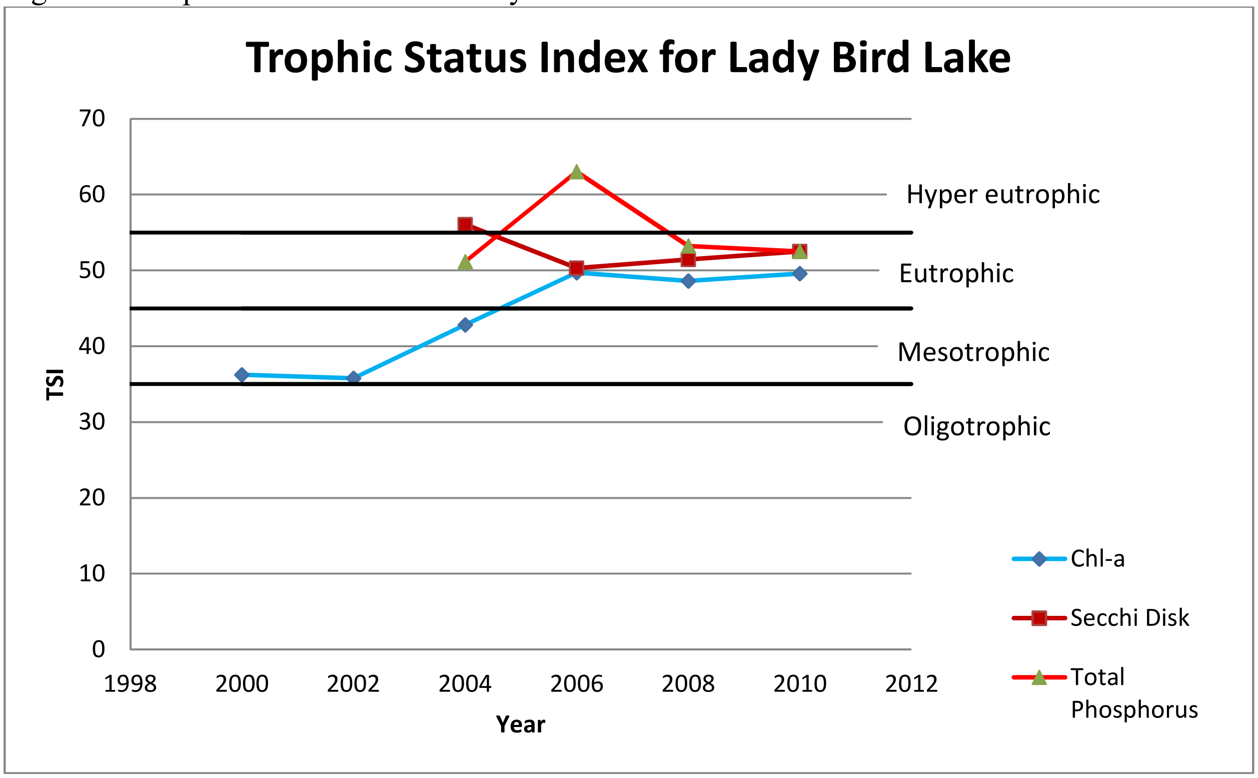
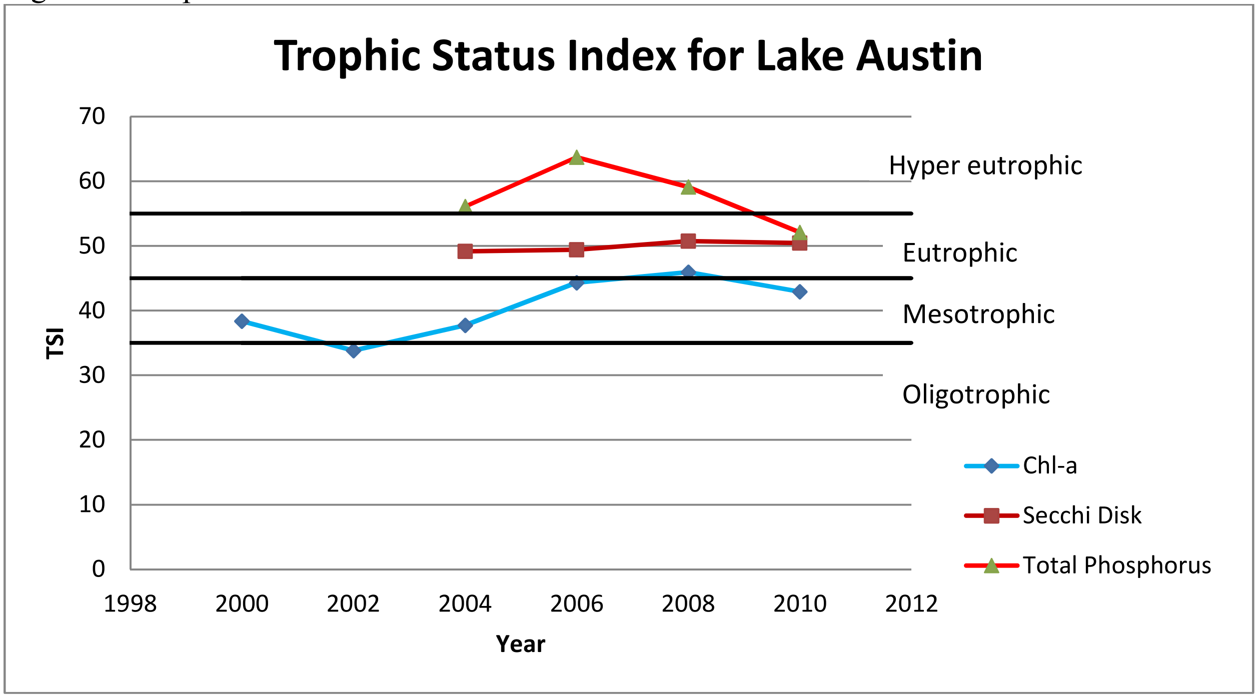


Figure 5: Trophic Status Index for Lake Austin



<u>Phytoplankton Trends</u>

Regression analysis of transformed blue-green counts at the Davis WTP using two degree polynomial B-splines with 153 breakpoints fit the data moderately well (R^2=0.7282). However, the model tended to underestimate the blue-green counts when the values were large (Figure 11). A full list of model results can be seen in Appendix B. Minor increases and decreases were predicted in 1993, 1996, 1998, 2001, 2002, and the end of 2004. More substantial increases and decreases in the blue-green algae counts were predicted in 2007, 2009, 2010, 2011, 2012, and 2013. This pattern would indicate an increase in blue-green algae intensity in the last five years at the Davis WTP.

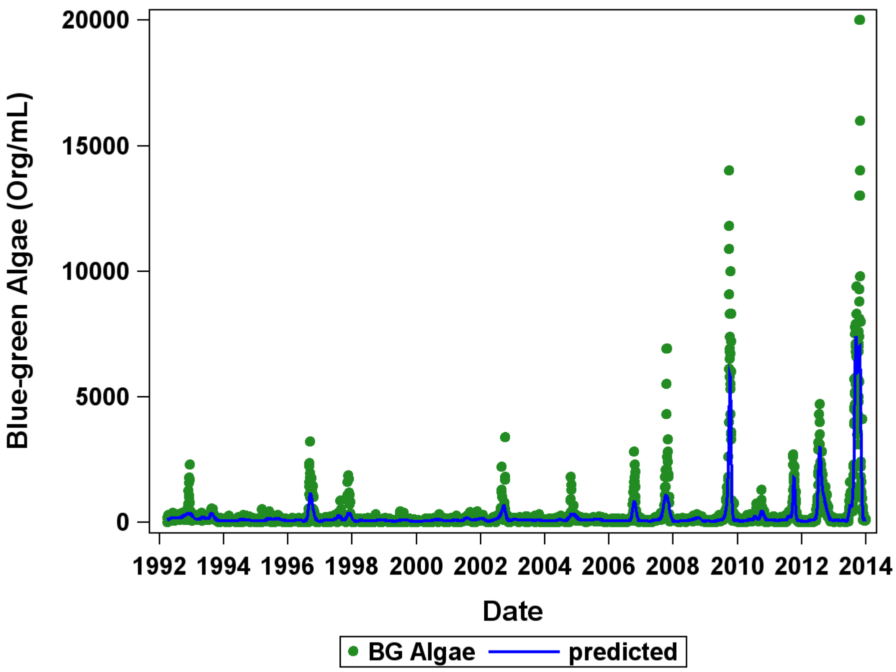


Figure 11: Regression model using the two degree polynomial B-spline of time with 153 breakpoints at the Davis WTP on Lake Austin.

To obtain a more generalized inference of a trend in counts over time, the number of breakpoints used in the B-spline was decreased to five which included 29 April 1996, 28 February 2001, 29 June 2005, 21 August 2009, and 21 March 2012. Slope estimates and confidence intervals for the modeled quantile regression equations using these breakpoints for the Davis WTP can be seen in Table 12. A positive slope indicates that the blue-green algae counts are increasing during a time period while a negative slope would indicate that the blue-green algae counts are decreasing during a time period. Change in blue-green algae counts at the Davis WTP was negligible for the median, 75th percentile, and 80th percentile regression equations until 21 March 2012. At which time, the blue-green algae counts for these percentiles increased (Figure 12). In the 95th percentile, the blue-green algae counts increased from 2005 to 2009. There was a decrease in counts from 2009 to 2012, but in 2012 the blue-green counts increased once again at this percentile.

Table 14: Slope estimates and confidence intervals for the quantile regression analysis of blue-green algae counts at the Davis WTP using a two degree polynomial B-spline with five breakpoints.

Quantile Regression	Date Range	Slope Estimate	95% Confidence Interval
Median (Q50)	01APR1992 – 29APR1996	-0.0004	-0.0006 to -0.0002
	29APR1996 – 28FEB2001	0.0004	0.0002 to 0.0007
	28FEB2001 – 29JUN2005	0	-0.0001 to 0.0002
	29JUN2005 – 21AUG2009	0	0 to 0.0004
	21AUG2009 – 21MAR2012	0	-0.0005 to 0
	21MAR2012 – 31DEC2013	**0.0036**	**0.0029 to 0.0037**
75th percentile (Q75)	01APR1992 – 29APR1996	-0.0004	-0.0004 to -0.0003
	29APR1996 – 28FEB2001	0.0003	0.0003 to 0.0004
	28FEB2001 – 29JUN2005	0	-0.0001 to 0
	29JUN2005 – 21AUG2009	0.0003	0.0003 to 0.0005
	21AUG2009 – 21MAR2012	-0.0002	-0.0007 to 0.0001
	21MAR2012 – 31DEC2013	**0.0053**	**0.0047 to 0.0062**
80th percentile (Q80)	01APR1992 – 29APR1996	-0.0003	-0.0005 to -0.0003
	29APR1996 – 28FEB2001	0.0003	0.0002 to 0.0006
	28FEB2001 – 29JUN2005	0	-0.0003 to 0
	29JUN2005 – 21AUG2009	0.0004	0.0002 to 0.0007
	21AUG2009 – 21MAR2012	-0.0002	-0.001 to 0.0002
	21MAR2012 – 31DEC2013	**0.005**	**0.0042 to 0.006**
95th percentile (Q95)	01APR1992 – 29APR1996	0.0009	0.0002 to 0.001
	29APR1996 – 28FEB2001	-0.0017	-0.002 to -0.0006
	28FEB2001 – 29JUN2005	**0.0011**	**0 to 0.0016**
	29JUN2005 – 21AUG2009	**0.0015**	**0.0001 to 0.0022**
	21AUG2009 – 21MAR2012	-0.0035	-0.0041 to -0.002
	21MAR2012 – 31DEC2013	**0.0057**	**0.0045 to 0.0065**

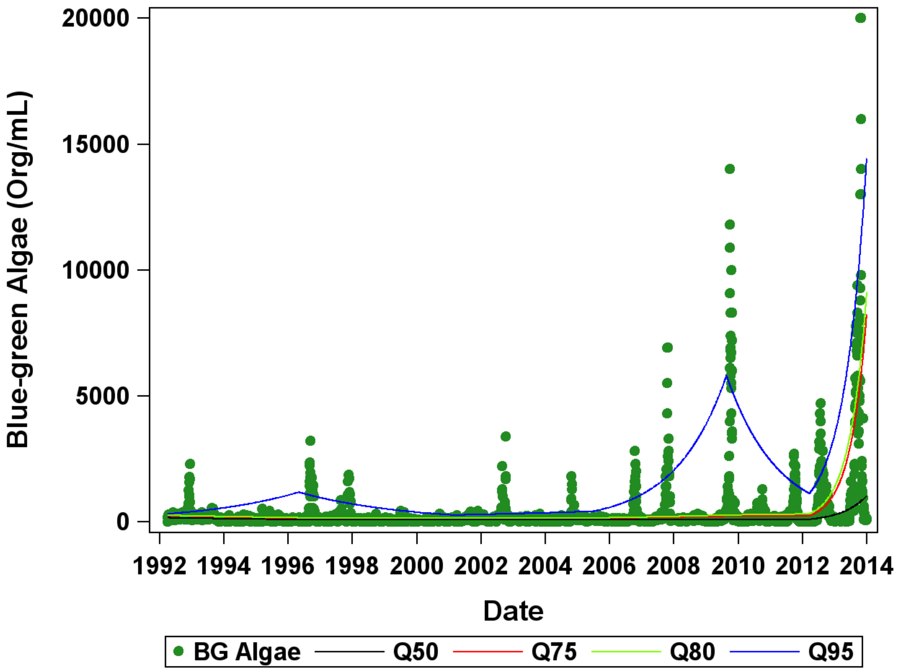


Figure 12: Blue-green algae counts with the median (Q50), 75th percentile (Q75), 80th percentile (Q80), and 95th percentile (Q95) piecewise regressions using five breakpoints at the Davis WTP on Lake Austin.

The 95th percentile was able to distinguish a change in algal bloom intensity easier than the other percentiles because it focuses on the extreme data points. However, the other regions of the data (median, 75th percentile, 80th percentile) did not indicate changes in algal counts prior to 2012, which coincides with previous analysis for blue-green algae trends at the Davis WTP (Richter 2010). Thus blue-green bloom intensity increased beginning in 2005 until 2009 when blue-green counts were higher at the Davis WTP then previously recorded. The brief decline in blue-green algae counts in 2010 and 2011 signifies that conditions do not exist every year at the Davis WTP for blue-green algae to respond like in 2009; however, maximum blue-green counts were still higher in 2010-11 than in many previous years. After 21 March 2012, all tested quantile regressions showed an increase in blue-green algae at the Davis WTP. Thus, not only are the most extreme data counts getting higher but more days where algae counts are collected throughout the year are containing higher blue-green counts. This indicates not only an increase in bloom intensity but an increase in bloom frequency or duration, something that was only speculative in previous lake reports (Richter 2010).

The increased frequency or duration of the high blue-green counts may be partially related to the altered water management within the Colorado River. Large average daily flows (>20,000 cfs) have not been observed at Tom Miller Dam since 2007 and the average daily flow was under 1,000 cfs in 2012-13, with the exception of three data points (Figure 13). The lower flows coincide with the recent high blue-green algae counts. High flows through Tom Miller Dam have typically been experienced during the months of April to October, which is a time of year when algal blooms are possible due to the warm temperatures. With the lack of higher flows in the summer months, the blue-green counts since 2008 have been higher in July, August, and September (Figure 14). Blue-green counts collected before 2008, were higher during September, October, and November. October continues to be the month with the highest blue-green algae counts while the counts in December seem to have decreased in intensity.

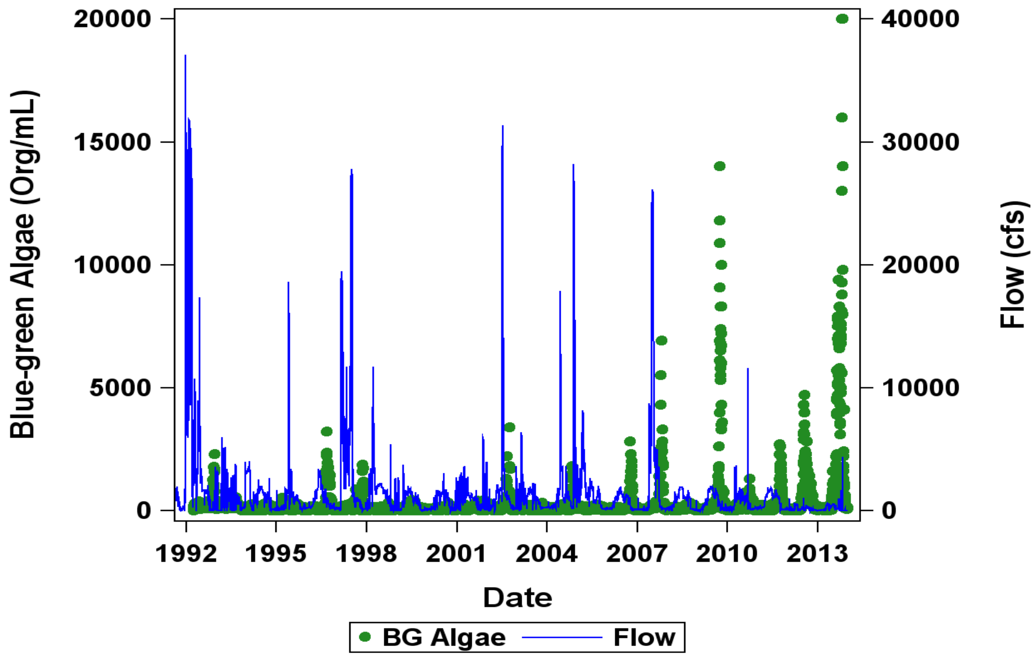


Figure 13: Blue-green algae counts (organisms/mL) from the Davis WTP and flow (cfs) from Tom Miller Dam.

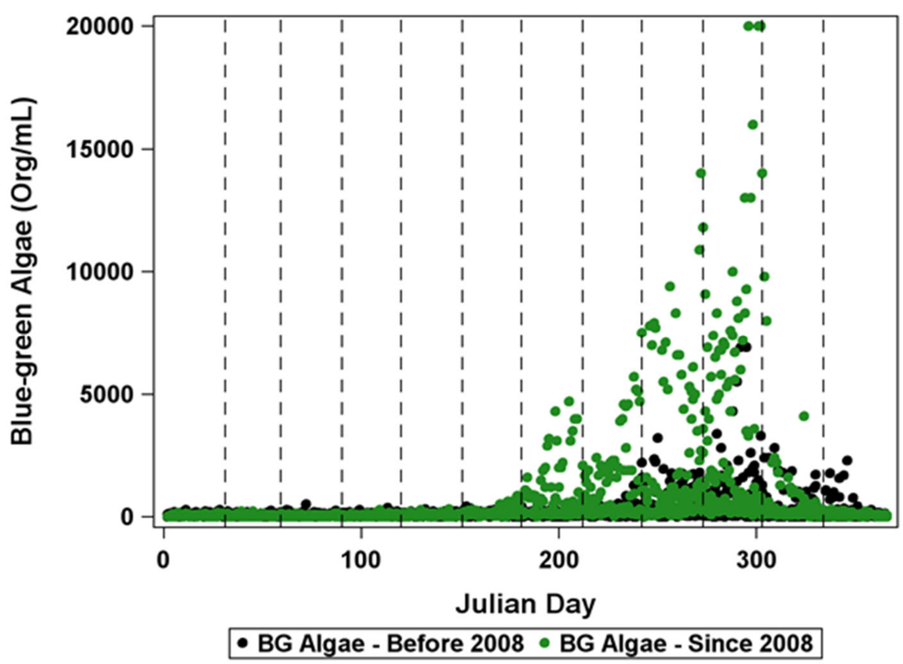


Figure 14: Blue-green algae counts (organisms/mL) at the Davis WTP collected prior to 01January2008 (black) and since 01January2008 (green). Dashed vertical lines separate the graph into 12 months.

Correlation analysis (Pearson) was used to compare the natural log of the blue-green algae counts collected at the Davis WTP against flow for the time period before 2008 and since 2008 by month of the year. Results indicated that there was no strong significant linear relationship between the transformed algae counts and flow prior to 2008 in any month; however, there were significant relationships present in June, July, August, and September in data collected since 2008 (Table 13). Each of these relationships was negative indicating higher blue-green counts when there was less flow through Tom Miller Dam in these months. This helps to strengthen the argument that the lack of higher flows in the summer months through Lake Austin at least partially explains a longer duration or higher frequency of blue-green blooms. Raw counts of blue-green algae compared to flow can be seen in Figure 15 for July, August, September, and October. High flows through Tom Miller Dam were hardly ever observed and the dominance of a low flow during this month seems to be a good condition that promotes higher blue-green counts; however, it is not the only prerequisite because not all data during the month of October supports high counts. Thus the flow is only considered as a partial component contributing to blue-green algal abundance.

Table 15: Pearson correlation coefficients between blue-green algae counts (log transformed) collected at the Davis WTP and flow at Tom Miller Dam before 2008 and since 2008.

	Prior to 2008		Since 2008	
Month	N	Correlation Coefficient	N	Correlation Coefficient
January	141	0.16	95	0.14
February	132	0.14	88	0.29
March	191	0.14	109	-0.02
April	217	0.21	113	0.13
May	199	0.19	113	0.12
June	206	0.23	112	-0.49*
July	182	0.08	107	-0.84*
August	213	0.12	117	-0.77*
September	206	-0.11	113	-0.58*
October	212	-0.09	128	0.08
November	179	0.14	114	-0.23
December	188	0.14	123	-0.04

*correlation with p-value < 0.0001

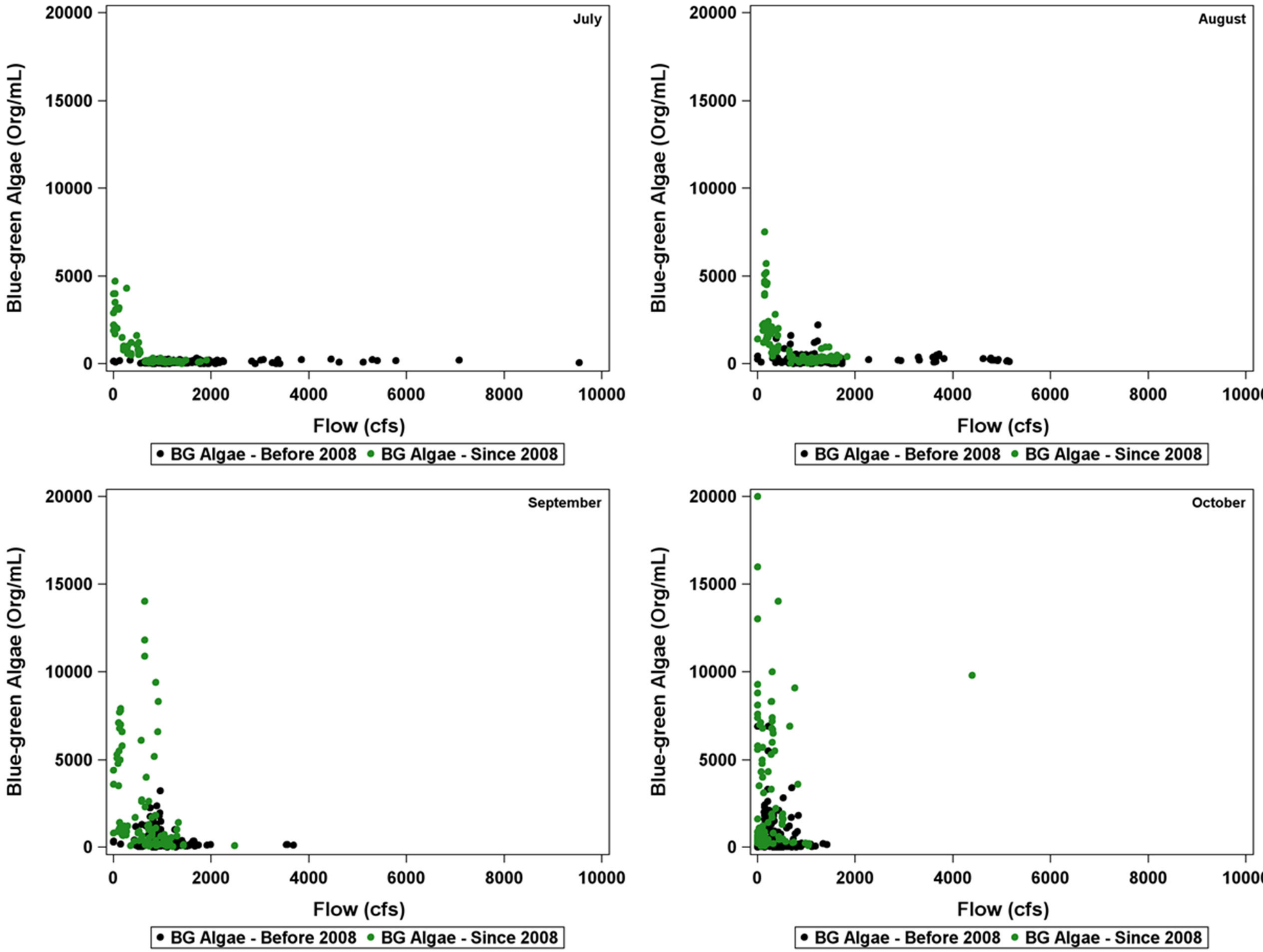


Figure 15: Blue-green algae counts (organisms/mL) at the Davis WTP collected prior to 01January2008 (black) and since 01January2008 (green) in relation to the flow at Tom Miller Dam during the month of July (top-left), August (top-right), September (bottom-left), and October (bottom-right).

Similar to the Daivs WTP, the model of blue-green algae counts using two degree polynomial B-splines with 153 breakpoints fit the data well at the Ullrich WTP (R^2=0.7579) but the model tended to underestimate the blue-green counts when values became large (Figure 16). The full model output for each slope estimate can be seen in Appendix C. Minor increases and decreases in blue-green algae counts were predicted in 1996 and every year after 2000, with substantial increases and decreases occurring after 2009. The pattern matches a similar pattern of increased blue-green intensity in the last five years which was noted at the Davis WTP.

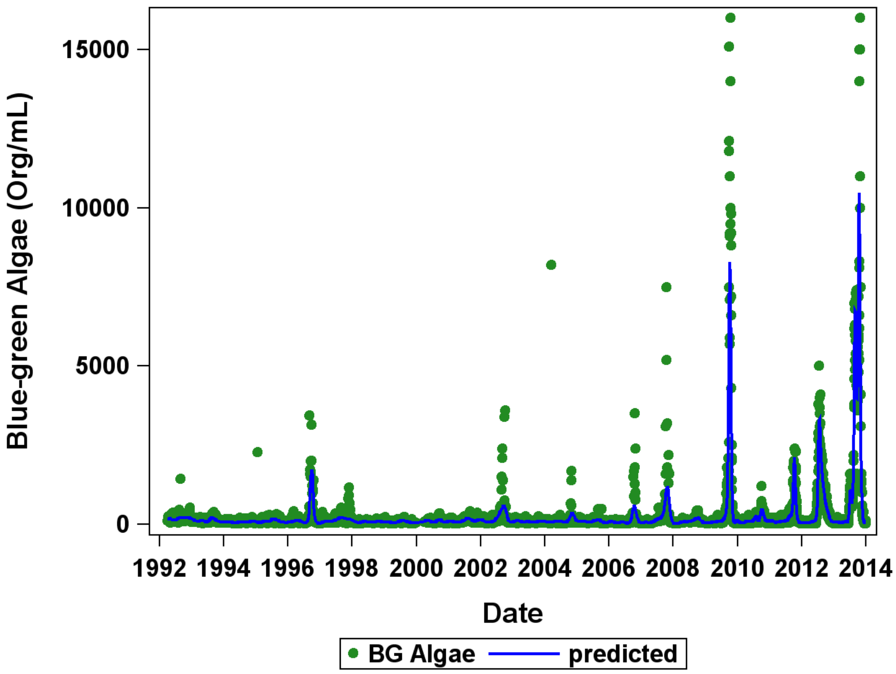


Figure 16: Regression model using the two degree polynomial B-spline of time with 153 breakpoints at the Ullrich WTP on Lake Austin.

For the Ullrich WTP, there was not as many algal blooms noted in the earlier time periods and a smoothing function with four breakpoints fit the data better than five breakpoints. The four breakpoints consisted of 09 September 1997, 12 February 2004, 26 October 2009, and 17 April 2012. Slope estimates and confidence intervals for the modeled quantile regression equations using these breakpoints can be seen in Table 14. A positive slope indicates that the blue-green algae counts are increasing during a time period while a negative slope indicates that the blue-green algae counts are decreasing during a time period.

Table 16: Slope estimates and confidence intervals for the quantile regression analysis of blue-green algae counts at the Ullrich WTP using a two degree polynomial B-spline with four breakpoints.

Quantile Regression	Date Range	Slope Estimate	95% Confidence Interval
Median (Q50)	01APR1992 – 09SEPT1997	-0.0001	-0.0002 to -0.0001
	09SEPT1997 – 12FEB2004	0.0001	0.0001 to 0.0002
	12FEB2004 – 26OCT2009	0	-0.0001 to 0.0002
	26OCT2009 – 17APR2012	-0.0001	-0.0004 to 0.0002
	17APR2012 – 31DEC2013	**0.0041**	**0.0032 to 0.0048**
75^{th} percentile (Q75)	01APR1992 – 09SEPT1997	-0.0001	-0.0002 to 0
	09SEPT1997 – 12FEB2004	0.0001	-0.0001 to 0.0002
	12FEB2004 – 26OCT2009	0.0002	0.0001 to 0.0004
	26OCT2009 – 17APR2012	0	-0.0006 to 0.0006
	17APR2012 – 31DEC2013	**0.0051**	**0.0039 to 0.0065**
80^{th} percentile (Q80)	01APR1992 – 09SEPT1997	-0.0001	-0.0002 to -0.0001
	09SEPT1997 – 12FEB2004	0.0001	0 to 0.0002
	12FEB2004 – 26OCT2009	0.0003	0.0001 to 0.0005
	26OCT2009 – 17APR2012	0.0001	-0.0005 to 0.0005
	17APR2012 – 31DEC2013	**0.0047**	**0.0036 to 0.0056**
95^{th} percentile (Q95)	01APR1992 – 09SEPT1997	0.0002	0.0001 to 0.0004
	09SEPT1997 – 12FEB2004	-0.0005	-0.0009 to -0.0002
	12FEB2004 – 26OCT2009	**0.0017**	**0.0008 to 0.0021**
	26OCT2009 – 17APR2012	-0.0028	-0.0036 to -0.0016
	17APR2012 – 31DEC2013	**0.0049**	**0.0043 to 0.0056**

Change in blue-green algae counts at the Ullrich WTP was negligible for the median, 75^{th} percentile, and 80^{th} percentile regression equations until the 2012 breakpoint. In 2012, the blue-green algae counts for these percentiles increased (Figure 17). In the 95^{th} percentile, the blue-green algae counts increased from 2004 to 2009. Thus the algae bloom intensity increased during this time period at the Ullrich WTP, which was a similar trend to increased intensity at the Davis WTP. The counts in 2010-12 were lower when compared to the counts in 2009, which is the reason for the decrease in counts noted from the model, but blue-green algae counts observed during this time frame were still higher than in many years prior to 2009. The increase in blue-green algae counts from 2012 to 2013 for all quantiles indicates that the blue-green counts are getting higher and that more data points contain high values. Thus, the blue-green algae bloom is increasing in intensity and frequency or duration at the Ullrich WTP.

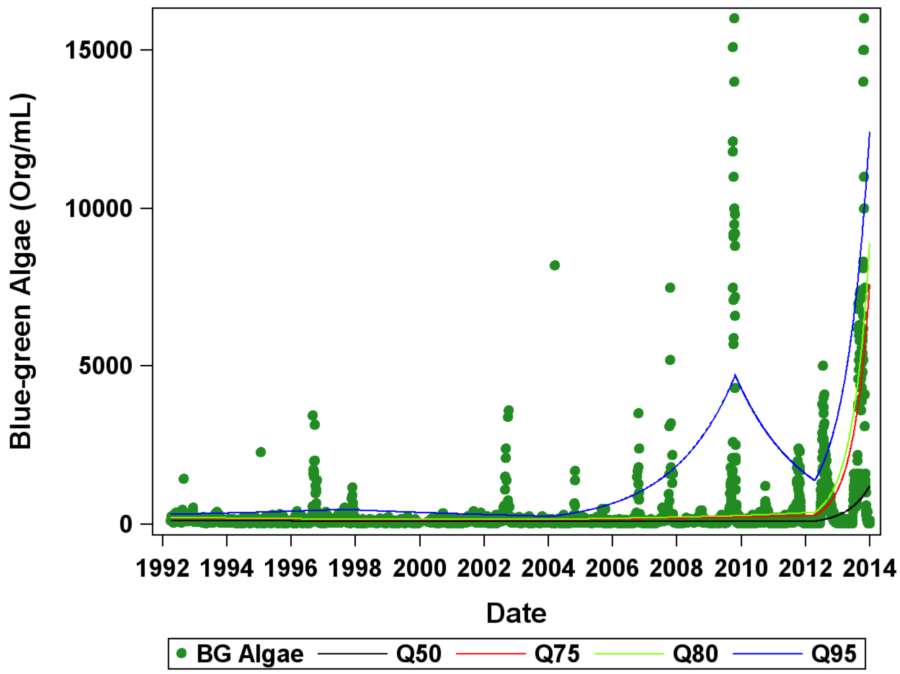


Figure 17: Blue-green algae counts with the median (Q50), 75th percentile (Q75), 80th percentile (Q80), and 95th percentile (Q95) piecewise regressions using four breakpoints at the Ullrich WTP on Lake Austin.

Higher blue-green algae counts have been observed at the Ullrich WTP from July to November since the beginning of 2008 (Figure 18). The increased frequency or duration of blue-green algae blooms at the Ullrich WTP can be attributed to higher counts observed in July and August as high counts prior to 2008 were observed from September to November. Blue-green counts have always been highest during October.

Like at the Davis WTP, the higher counts in July and August at the Ullrich WTP can be partially attributed to the lack of higher flow through Tom Miller Dam during the summer months. Pearson correlation analysis results indicated that there was no strong significant linear relationship between the log transformed blue-green algae counts and flow prior to 2008 in any month; however, there were significant relationships present in June, July, August, and September in data collected since 2008 (Table 15). The strong relationships found in these months were all negative indicating higher blue-green counts when there was less flow through Tom Miller Dam in these months. Raw counts of blue-green algae compared to flow can be seen in Figure 19 for July, August, September, and October. There are some low blue-green counts present in low flow conditions thus the flow is only considered as a partial component contributing to blue-green algal abundance.

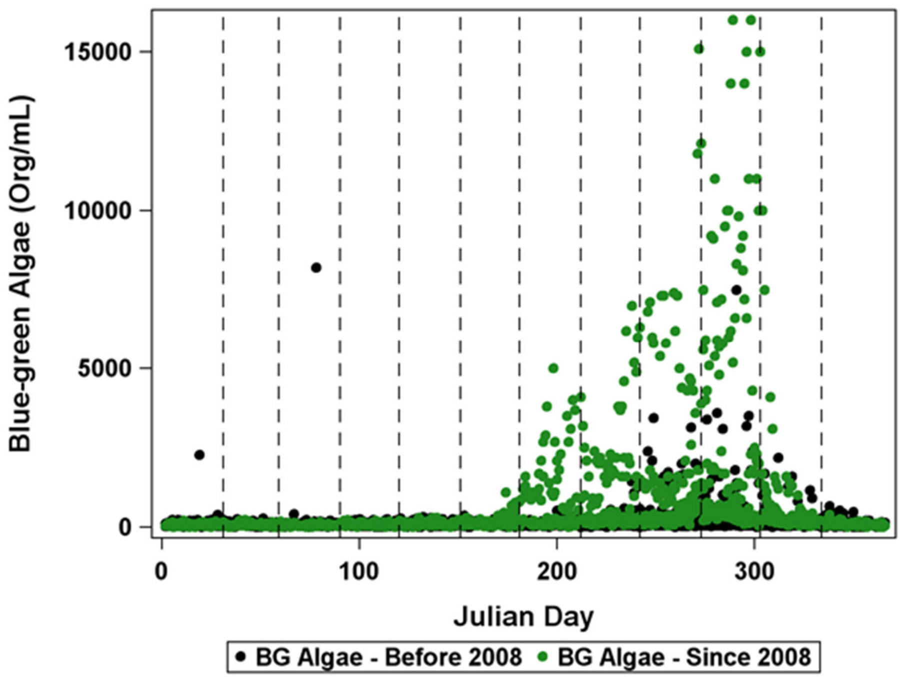


Figure 18: Blue-green algae counts (organisms/mL) at the Ullrich WTP collected prior to 01January2008 (black) and since 01January2008 (green). Dashed vertical lines separate the graph into 12 months.

Table 17: Pearson correlation coefficients between blue-green algae counts (log transformed) collected at the Ullrich WTP and flow at Tom Miller Dam before 2008 and since 2008.

	Prior to 2008		Since 2008	
Month	N	Correlation Coefficient	N	Correlation Coefficient
January	112	0.11	92	-0.16
February	110	0.10	83	0.02
March	136	0.10	104	0.05
April	157	0.22	103	0.15
May	152	0.20	100	-0.03
June	147	0.21	101	-0.39*
July	130	0.32	99	-0.81*
August	148	0.14	106	-0.83*
September	143	-0.05	107	-0.52*
October	146	-0.02	119	0.13
November	124	0.11	105	-0.18
December	128	0.10	113	-0.03

*correlation with p-value < 0.0001

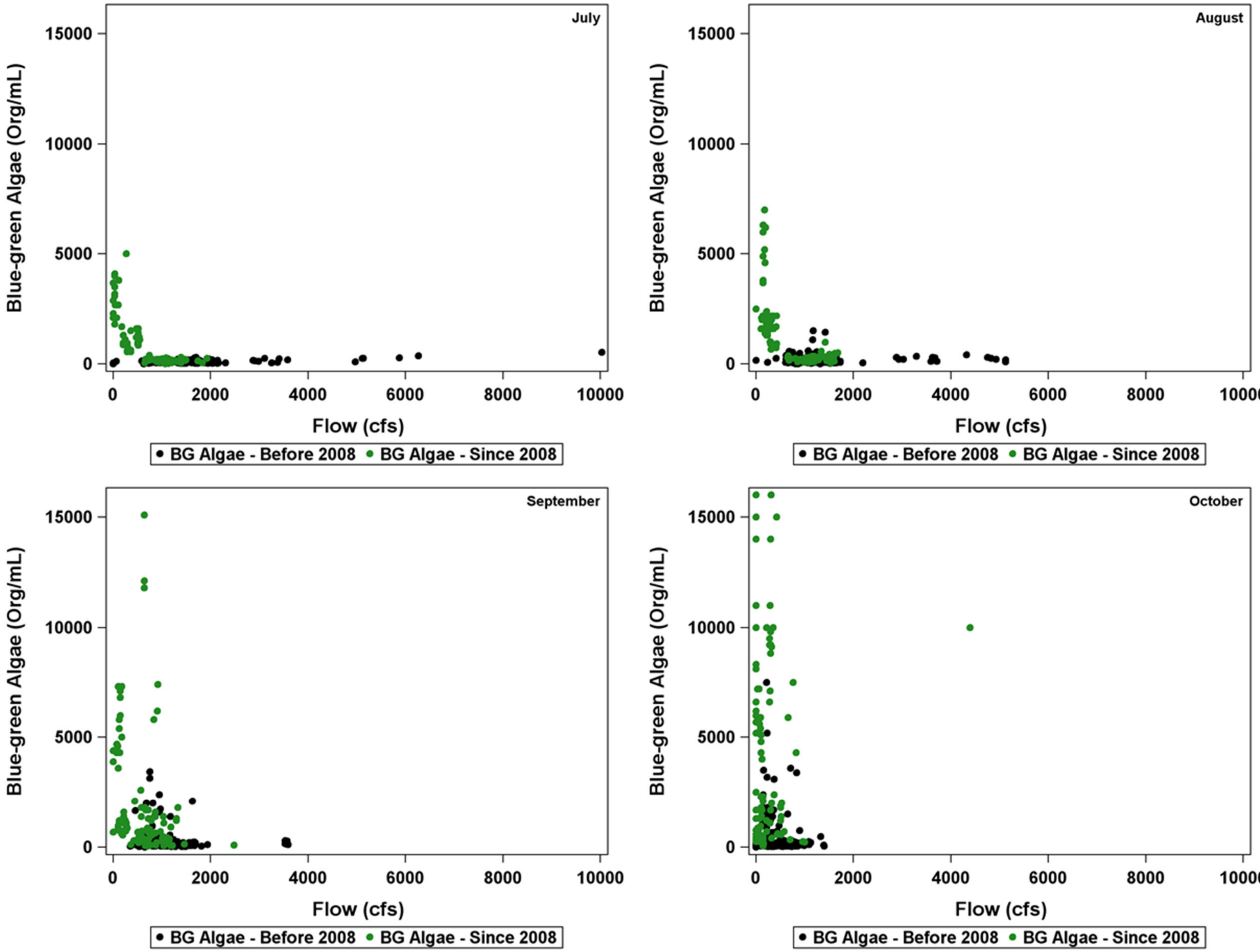


Figure 19: Blue-green algae counts (organisms/mL) at the Ullrich WTP collected prior to 01January2008 (black) and since 01January2008 (green) in relation to the flow at Tom Miller Dam during the month of July (top-left), August (top-right), September (bottom-left), and October (bottom-right).

Further confirmation of increased bloom frequency or duration was examined using logistic regression. Blue-green algae counts were designated as either bloom or non-bloom based on the count compared to a bloom threshold. The year was used as an explanatory variable in the logistic regression. In order to determine if the logistic model fit the data well, it was determined how well the model correctly classified bloom versus non-bloom counts. The *c-statistic,* the area under a receiver operating characteristic (ROC) curve, was used to determine if the model correctly classified each algal count as a bloom or non-bloom. The *c-statistic* for the logistic model at the Davis WTP was 0.794 (Figure 20). The ROC curve rises relatively quickly for the model; however, there are a number of false positives predicted in years with a low probability of a bloom. Adding other parameters to the model would probably increase the rise in the ROC curve but with this simple model we should be able to determine the probability of a bloom occurring on a given day within a year.

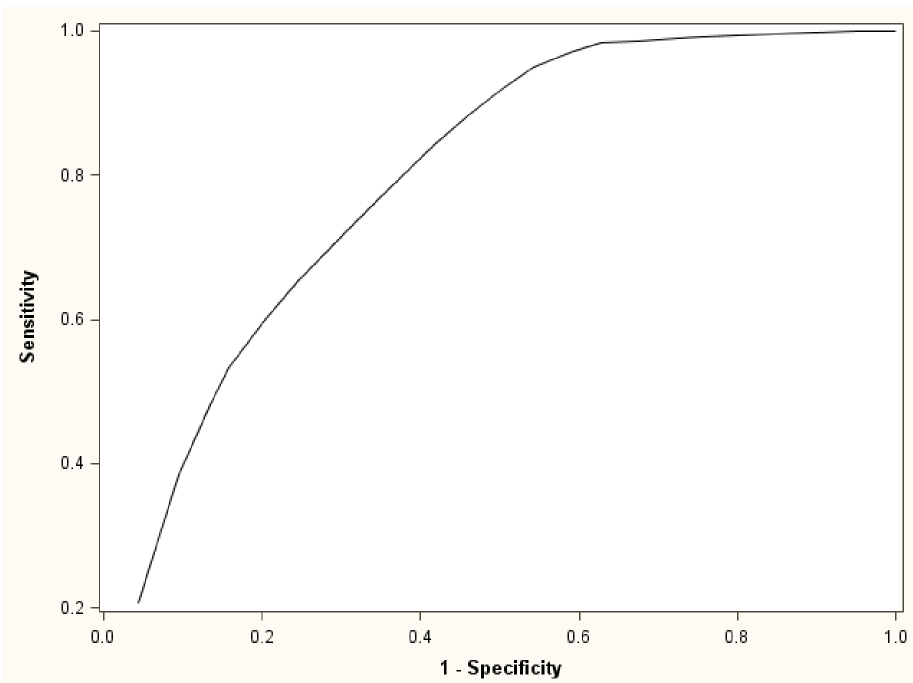


Figure 20: Receiver Operating Characteristic (ROC) curve for the logistic model of algal blooms with year as a predictor variable at the Davis WTP. Area under the curve is 0.794 (*c-statistic*).

Output from a logistic regression performed on blue-green algae data collected at the Davis WTP showed that the probability of the occurrence of a bloom was significantly higher in 2013 than any other year except 2012 (Table 16). The probability of the occurrence of a bloom in the years 2009 and 2012 were not different from each other but were significantly higher than past years except that 2009 was not significantly different from the predicted probability in 1992. The probabilities of blooms in 1992, 1996, and 1997 were also significantly higher than many other years but not all other years. Full results for tests of significant differences between the probability of an algae bloom between years are found in Appendix D.

Table 18: Probability of a blue-green algae bloom at the Davis WTP on Lake Austin by year.

Year	Probability of Bloom	95% Confidence Interval	
		Lower Limit	Upper Limit
1992	0.269	0.193	0.361
1993	0.118	0.075	0.181
1994	0.009	0.002	0.045
1995	0.015	0.004	0.051
1996	0.206	0.152	0.274
1997	0.148	0.100	0.214
1998	0.011	0.002	0.054
1999	0.017	0.005	0.058
2000	0.015	0.003	0.069
2001	0.056	0.028	0.111
2002	0.163	0.112	0.232
2003	0.010	0.002	0.048
2004	0.077	0.044	0.133
2005	0.003	0.000	0.051
2006	0.128	0.084	0.192
2007	0.184	0.130	0.253
2008	0.009	0.002	0.044
2009	0.301	0.238	0.372
2010	0.166	0.125	0.217
2011	0.153	0.114	0.204
2012	0.380	0.322	0.442
2013	0.444	0.384	0.506

The *c-statistic* for the logistic model at the Ullrich WTP was 0.793 (Figure 21). Like the Davis WTP ROC curve, the sensitivity rises relatively quickly for the model and while not optimized for prediction, this simple model can be used to determine the probability of a bloom occurring on a given day within a year at the Ullrich WTP. Output from a logistic regression performed on blue-green algae data collected at the Ullrich WTP showed that the probability of the occurrence of a bloom was significantly highest in 2009, 2012, and 2013 (Table 17). Other years with statistically higher probabilities include 1992, 1996, 1997, 2002, 2007, 2011, and 2012 (Appendix E). It could be inferred that the occurrence of a blue-green algae bloom has become more probable in recent years and that the frequency/duration of bloom periods on Lake Austin has increased because the probabilities have been high from 2009-2013 while prior to these years only five out of 17 years had high probabilities of a blue-green algal bloom.

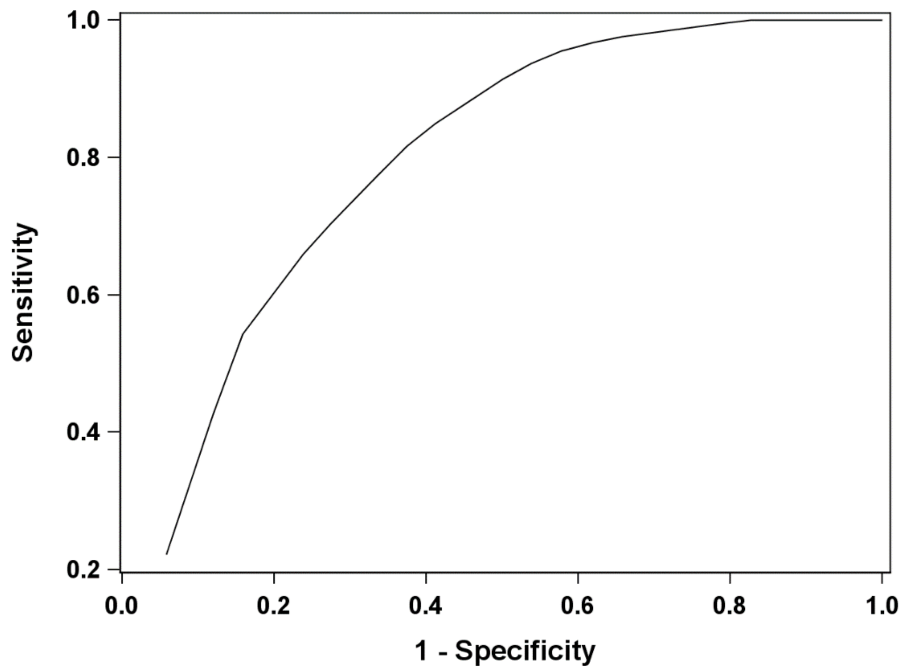


Figure 21: Receiver Operating Characteristic (ROC) curve for the logistic model of algal blooms with year as a predictor variable at the Ullrich WTP. Area under the curve is 0.85 (*c-statistic*).

Table 19: Probability of a blue-green algae bloom at the Ullrich WTP on Lake Austin by year.

Year	Probability of Bloom	95% Confidence Interval	
		Lower Limit	Upper Limit
1992	0.192	0.122	0.289
1993	0.058	0.027	0.120
1994	0.004	0.000	0.065
1995	0.035	0.014	0.086
1996	0.185	0.124	0.267
1997	0.156	0.098	0.238
1998	0.005	0.000	0.075
1999	0.005	0.000	0.071
2000	0.030	0.009	0.100
2001	0.046	0.018	0.112
2002	0.189	0.125	0.277
2003	0.005	0.000	0.076
2004	0.090	0.048	0.163
2005	0.033	0.011	0.090
2006	0.111	0.063	0.187
2007	0.197	0.133	0.283
2008	0.032	0.011	0.087
2009	0.378	0.305	0.458
2010	0.130	0.094	0.178
2011	0.229	0.181	0.285
2012	0.400	0.341	0.462
2013	0.436	0.376	0.498

In order to examine the complex system of phytoplankton growth, a structural equation model analysis was done testing the model where flow through Lake Austin was impacted by the previous day flow; phosphate concentrations were affected by flow through Lake Austin and previous day phytoplankton counts; and phytoplankton counts were affected by flow through Lake Austin, previous day phytoplankton counts, and phosphate concentrations. Total phytoplankton and blue-green algae counts were used in separate models as the dependent variables for each water treatment plant. Each of the four models was run for 90 imputed data sets and the fit statistics were combined for all 90 model outputs (Table 18). The 95% confidence interval is shown as well as the combined mean for each statistic so that the range of values for each fit statistic can be seen. The first fit statistic was to compare the covariance matrix implied by the models to the sample covariance matrices. The 'Model Chi-Square p-value' showed that in the majority of the models the covariance matrix implied by the model was close enough to the sample covariance matrix that the chi-square test failed to show that the matrices were different (p-value >> 0.05). This means that each model was supported by the data and further examination of each model was warranted. The goodness of fit index (GFI) and Bentler-Bonett normed fit index (NFI) measures fit of a model on a scale from 0 to 1 with 1.0 indicating the best fit. The 95% confidence interval for both fit indices in each model was above 0.9994, indicating that the model fit the data well. A value of 0.95 for these fit statistics can be used in some instances to declare a model fits the data well. As a final test for model fit, the root mean square error of approximation (RMSEA) was examined. Browne and Cudeck (1993) suggested that a RMSEA < 0.05 indicates good fit of the model. The highest mean RMSEA for the current models was 0.0094 which is much lower than the threshold of 0.05, and in fact the highest upper 90% confidence limit for the RMSEA of the four models ranged from 0.0223 to 0.0245 which was much lower than this threshold as well. As the sample size and degrees of freedom grow larger, the RMSEA will get closer to 0 even if the model does not fit the data well, thus it is recommended that multiple fit statistics are used to determine goodness of fit for a model. Each fit statistic has its limitations but upon viewing all of the above fit statistics, the models fit the data well.

Table 20: Mean and 95% confidence intervals for fit statistics for model output from 90 imputed data sets using total phytoplankton and blue-green algae counts at both Davis and Ullrich WTP.

Phytoplankton/ Location	Fit Statistic	Mean	95% CI between models	
Total Phytoplankton/ Davis WTP	Model Chi-Square p-value (df = 3)	0.2181	0.1742	0.2620
	Goodness of Fit Index (GFI)	0.9997	0.9997	0.9997
	RMSEA	0.0094	0.0080	0.0108
	RMSEA lower 90% CL	0.0016	0.0008	0.0023
	RMSEA upper 90% CL	0.0234	0.0223	0.0245
	Bentler-Bonett NFI	0.9994	0.9994	0.9995
Total Phytoplankton/ Ullrich WTP	Model Chi-Square p-value (df = 3)	0.4124	0.3468	0.4780
	Goodness of Fit Index (GFI)	0.9998	0.9998	0.9998
	RMSEA	0.0057	0.0043	0.0071
	RMSEA lower 90% CL	0.0009	0.0004	0.0014
	RMSEA upper 90% CL	0.0189	0.0174	0.0204
	Bentler-Bonett NFI	0.9996	0.9996	0.9997
Blue-green algae/ Davis WTP	Model Chi-Square p-value (df = 3)	0.3472	0.2887	0.4057
	Goodness of Fit Index (GFI)	0.9998	0.9998	0.9998
	RMSEA	0.0064	0.0052	0.0076
	RMSEA lower 90% CL	0.0005	0.0002	0.0009
	RMSEA upper 90% CL	0.0199	0.0185	0.0212
	Bentler-Bonett NFI	0.9996	0.9995	0.9996
Blue-green algae/ Ullrich WTP	Model Chi-Square p-value (df = 3)	0.4304	0.3671	0.4936
	Goodness of Fit Index (GFI)	0.9998	0.9998	0.9999
	RMSEA	0.0050	0.0038	0.0063
	RMSEA lower 90% CL	0.0005	0.0002	0.0010
	RMSEA upper 90% CL	0.0185	0.0170	0.0199
	Bentler-Bonett NFI	0.9996	0.9996	0.9997

Parameter estimates, the variance for parameter estimates within each model, and the 95% confidence intervals for model results from the 90 imputed data sets can be viewed in Table 19 and 20. The flow equation in every model was the same and thus each model had the same parameter estimate for how the previous day flow impacted flow. There was also no imputed data for flow so the parameter estimate was the same in all 90 model outputs. In each model the result was trivial: flows were predicted to be high when the previous day flow was high.

Results for phosphate showed that the concentration was predicted to increase as flows and the previous day phytoplankton count increased. This was true for both total phytoplankton and blue-green algae at both the Davis and Ullrich WTP (Table 19 and 20). The parameter estimate for flow was relatively constant across locations and phytoplankton type. In each model the increase of flow was more influential on the phosphate concentrations than the previous day phytoplankton counts.

Results for the phytoplankton showed that the phytoplankton counts increased as the previous day phytoplankton counts increased, flow decreased, and phosphate concentrations increased. This was true for both total phytoplankton and blue-green algae at both the Davis and Ullrich WTP (Table 19 and 20). The parameter estimates for phosphate had more variance between the imputed model results but in every case the effect direction was the same. It appears that the phytoplankton at the Ullrich WTP experience larger changes in counts with changes in phosphate concentrations and the blue-green counts are impacted more than the total phytoplankton counts at each water treatment plant.

Table 21: Parameter estimates, within model estimate variance, and between imputed data set model 95% confidence intervals for phytoplankton structural equation model analysis.

Total Phytoplankton/Davis WTP												
Response Variable	Explanatory Variables											
	ln(Plankton) Lag			ln(Flow)			ln(Phosphate)			ln(Flow) Lag		
	Mean	95% CI between models		Mean	95% CI between models		Mean	95% CI between models		Mean	95% CI between models	
ln(Plankton)	0.1539	0.1517	0.1561	-0.0359	-0.0370	-0.0348	0.0382	0.0241	0.0522			
within model variance	0.0110	0.0110	0.0110	0.0028	0.0028	0.0028	0.0104	0.0103	0.0104			
ln(Phosphate)	0.0041	0.000623	0.00755	0.0550	0.0518	0.0581						
within model variance	0.0119	0.0118	0.0120	0.00299	0.00297	0.00301						
ln(Flow)										0.8366	0.8366	0.8366
within model variance										0.0062	0.0062	0.0062
Total Phytoplankton/Ullrich WTP												
ln(Plankton)	0.1376	0.1352	0.1399	-0.0301	-0.0314	-0.0287	0.0967	0.0771	0.1164			
within model variance	0.0109	0.0109	0.0110	0.00272	0.00271	0.00273	0.00916	0.00908	0.00925			
ln(Phosphate)	0.0174	0.0123	0.0225	0.0606	0.0569	0.0643						
within model variance	0.0134	0.0133	0.0135	0.00326	0.00324	0.00329						
ln(Flow)										0.8366	0.8366	0.8366
within model variance										0.0062	0.0062	0.0062

Table 22: Parameter estimates, within model estimate variance, and between imputed data set model 95% confidence intervals for phytoplankton structural equation model analysis.

Blue-green algae/Davis WTP												
Endogenous Variable	Explanatory Variables											
	ln(Plankton) Lag			ln(Flow)			ln(Phosphate)			ln(Flow) Lag		
	Mean	95% CI between models		Mean	95% CI between models		Mean	95% CI between models		Mean	95% CI between models	
ln(Plankton)	0.1525	0.1501	0.1548	-0.0206	-0.0228	-0.0185	0.1268	0.0979	0.1557			
within model variance	0.0111	0.0110	0.0111	0.00729	0.00727	0.00731	0.0270	0.0268	0.0272			
ln(Phosphate)	0.00390	0.00265	0.00515	0.0549	0.0517	0.0580						
within model variance	0.00460	0.00457	0.00462	0.00297	0.00295	0.00298						
ln(Flow)										0.8366	0.8366	0.8366
within model variance										0.0062	0.0062	0.0062
Blue-green algae/Ullrich WTP												
ln(Plankton)	0.1377	0.1355	0.1399	-0.0428	-0.0467	-0.0389	0.3063	0.2553	0.3573			
within model variance	0.0109	0.0109	0.0110	0.00747	0.00743	0.00751	0.0252	0.0250	0.0255			
ln(Phosphate)	0.00649	0.00468	0.00831	0.0603	0.0566	0.0640						
within model variance	0.00487	0.00483	0.00491	0.00325	0.00323	0.00328						
ln(Flow)										0.8366	0.8366	0.8366
within model variance										0.0062	0.0062	0.0062

In the model, as flow increases through the lake, influxes of phosphate were most likely causing the phosphate concentration to rise; however, the point source for the phosphate is unknown. The relationship between phytoplankton counts and the flow did not dispute the theory proposed by COA staff that the decreased flow through Lake Austin would increase the residence time in the lake and allow more phytoplankton to accumulate (total and blue-green). Even the relationship between the phytoplankton communities and the phosphate concentrations did not dispute staff thoughts as an increase in phosphate concentrations led to an increase in phytoplankton counts which led to a higher phosphate (maybe organic) concentration. It is the inner relation between all three parameters that begins to complicate interpretations and ultimately the model fails to explain this relationship well. When flows increase, the phosphate concentrations rise and the phytoplankton count should also rise according to the second equation in the model. However, according to the first equation the opposite occurs and the phytoplankton counts fall. Most likely this is due to the fact that very high flows simply wash the phytoplankton downstream. The model fails to incorporate this threshold aspect which is probably the main cause for low R^2 values in the phosphate and phytoplankton count equations ($R^2 < 0.1$).

Further investigation of the complex relationship between phytoplankton communities, flow regime, and nutrient concentrations is warranted. However, multiple similar models were created and an attempt was made to fit all of them to the data. In only two other scenarios was the model covariance matrix close enough to the sample covariance matrix to not be rejected by the model chi-square test. One model consisted of the addition of the previous day flow to the list of predictors for phytoplankton. The other scenario consisted of removing the previous day phytoplankton counts from the list of predictors for phosphate concentrations. In neither case did the model covariance matrix fit the sample covariance matrix as well as the above model.

Austin Lake Index Analysis

The Austin Lake Index (ALI) is calculated for Lady Bird Lake, Lake Austin, and Walter E. Long; however, this report pertains to Lady Bird Lake and Lake Austin so the data collected from Walter E. Long will not be presented. Numerical scores for the ALI and all sub-indices range from 0 to 100 with environmental quality categories ranging from very bad to excellent, respectively. The ALI for each lake has not changed much over time with the overall score falling in the fair category of environmental quality almost every year for each lake (Figure 22). This is not the case for all of the sub-indices as some of the sub-indices have changed over time. As the ALI is an average of the sub-indices, the lack of change in the overall score is likely due to an averaging effect where changes in sub-indices offset each other leading to a neutral impact on the overall score.

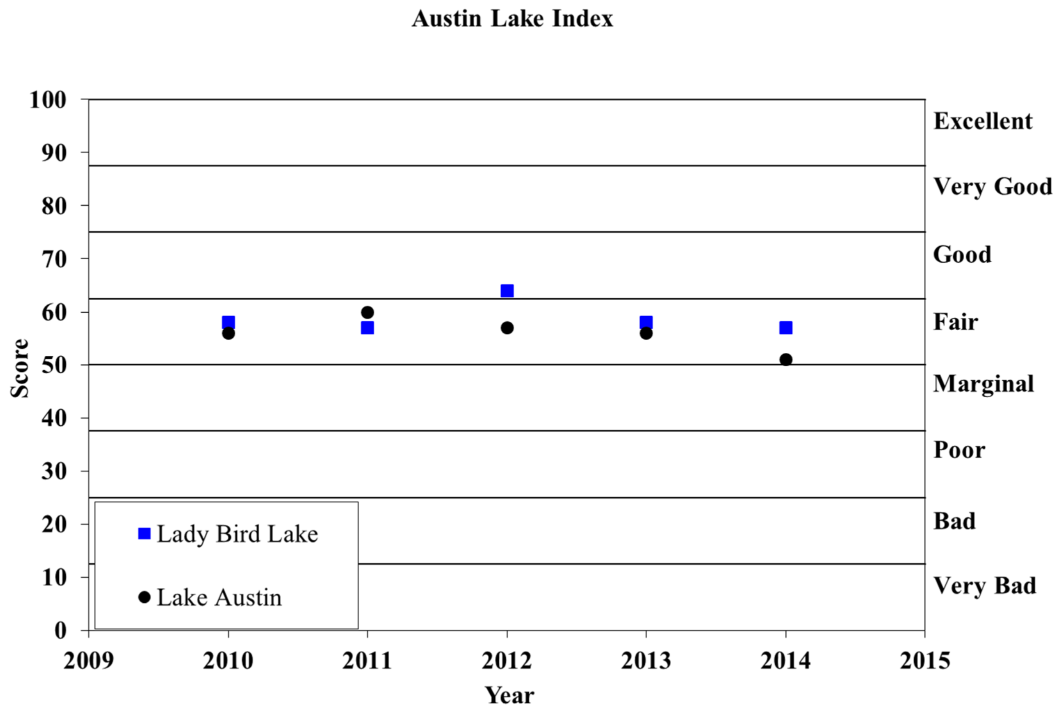


Figure 22: Austin Lake Index from 2010 to 2014.

The water quality sub-index of the ALI is a score based on concentrations of ammonia, nitrate, orthophosphorus, total suspended solids, conductivity, and *E. coli* bacteria counts. The water quality sub-index has typically been higher in Lake Austin due to the lower concentrations of total suspended solids when compared to Lady Bird Lake. However, the water quality sub-index for Lake Austin has decreased from 79 in 2011 to 66 in 2014 (Figure 23), primarily due to the total suspended solids component of the sub-index. This does not constitute a significant change over time in the actual total suspended solids concentration for Lake Austin but it does show that higher concentrations of total suspended solids have been collected in recent years. The water quality sub-index for Lady Bird Lake has decreased from 70 in 2011 to 49 in 2014 (Figure 23), primarily due to the nitrate component of the sub-index with contribution from the conductivity component in 2013-14. This would signify higher concentrations of nitrate and conductivity in recent years but cannot be interpreted as a significant increase in the raw concentrations of either nitrate or conductivity.

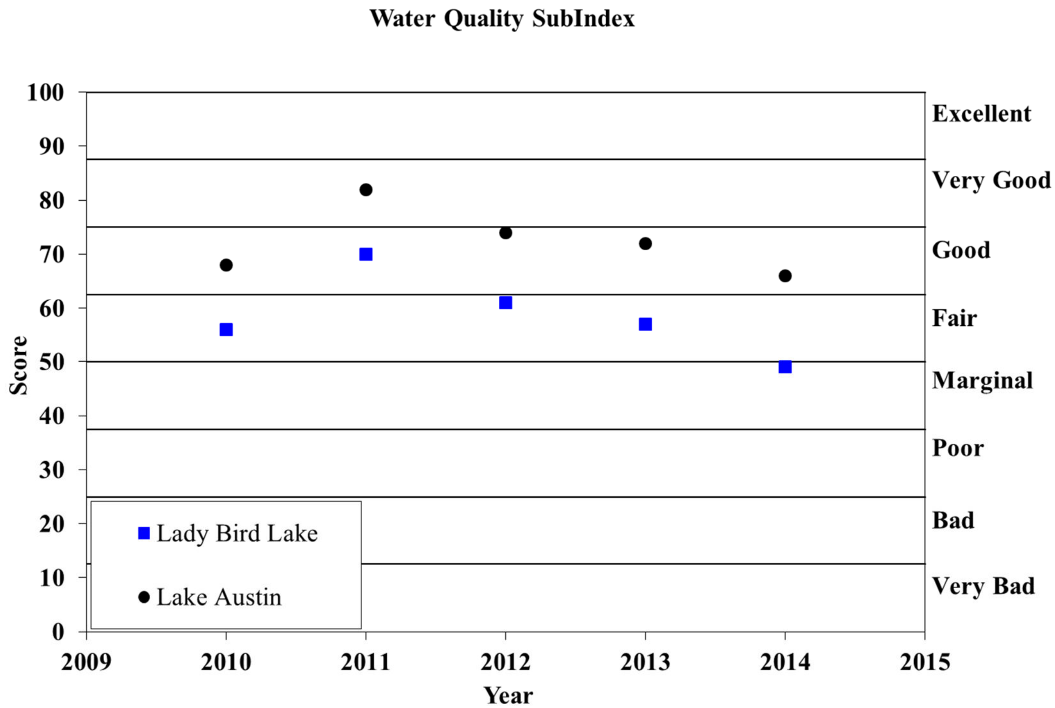


Figure 23: Water quality sub-index of the Austin Lake Index from 2010 to 2014.

The sediment quality sub-index of the ALI is a score based on concentrations of metals, herbicides/pesticides, and polycyclic aromatic hydrocarbons (PAH). The sediment quality sub-index has been consistently within the fair categorization in Lady Bird Lake but has varied from the very good classification to the fair classification in Lake Austin (Figure 24). The PAH component of the sediment quality sub-index is consistently higher within Lake Austin, indicating that higher concentrations of PAH are observed in the sediment of Lady Bird Lake. In addition, herbicides/pesticides have typically been detected in Lady Bird Lake sediment samples but have not been detected in Lake Austin sediment samples with the exception of 2014. This will also lead to a lower sediment quality sub-index for Lady Bird Lake. In 2014, concentrations of DDT (1,1,1-trichloro-2,2-bis(p-chlorophenyl)ethane) were detected in Lake Austin and Lady Bird Lake. DDT is a persistent insectide that has been shown to negatively impact the health of humans and animals at certain concentrations (ATSDR 2002, US EPA 2015). Currently, it is illegal to sell or distribute DDT in the United States. Lake Austin and Lady Bird Lake were not the only locations in Austin in 2014 where sediment contained detectable levels of DDT. Richter (2015) contains more in depth investigation of DDT in Austin watersheds through 2014.

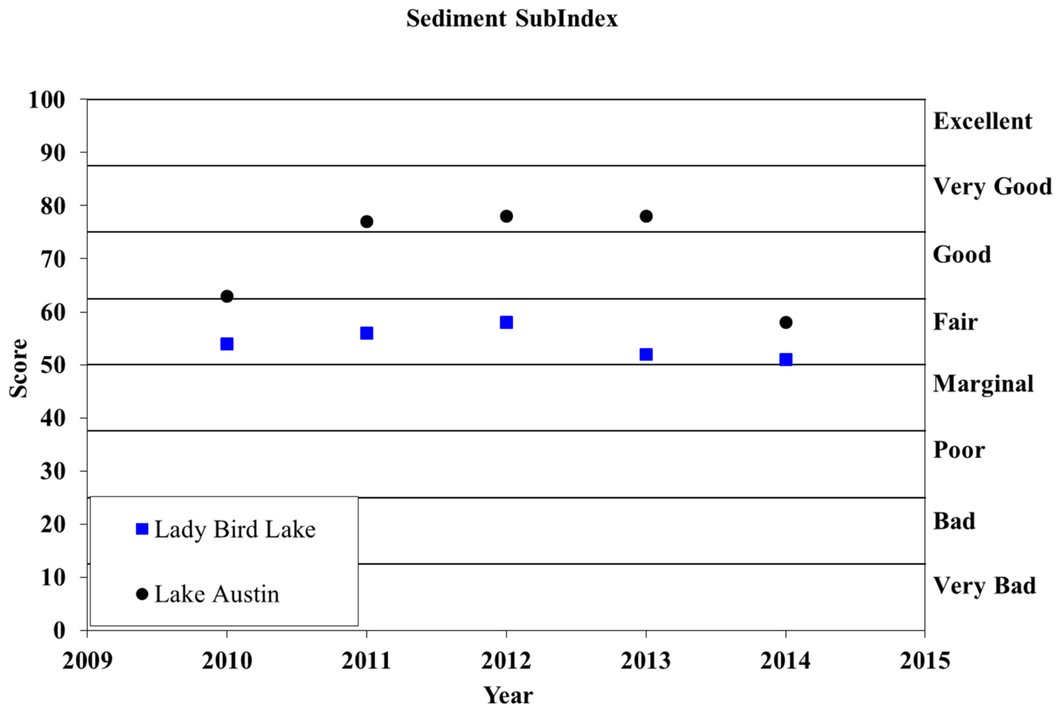


Figure 24: Sediment quality sub-index of the Austin Lake Index from 2010 to 2014.

The vegetation sub-index of the ALI is a score based on the total aquatic vegetative cover throughout the lake, the species richness of the vegetation, and the amount of vegetation that is non-native. Surveys are conducted by the Texas Parks and Wildlife Department (TPWD) each year on both lakes. Vegetation in Lady Bird Lake was not abundant in 2010 and 2011 but increased dramatically in 2012 when *Cabomba caroliniana* established in 31.74 ha (78.36 acres). The sub-index responded by increasing from 35 in 2011 to 84 in 2012 and has not fallen below 72 in the continuing years due to the on-going presence of *C. caroliniana* (Figure 25).

In Lake Austin, the vegetation sub-index has always been below 50 due to the fact that the majority of the vegetation has always consisted of *Hydrilla verticillata*, an exotic plant. Hydrilla has been a problematic species in Lake Austin since 1999, when it was documented as the dominant plant species covering 9.32 ha (23 acres, 1.4% of Lake Austin). In fact, the Lake Austin Hydrilla Task Force was formed in 2000 with members from the COA, Travis County, and Lower Colorado River Authority (LCRA). The goal of the task force was to develop a management plan for hydrilla. With cooperation from TPWD, *Ctenopharyngodon idella* (grass carp) were eventually stocked into Lake Austin and the cover of hydrilla slowly decreased. In the fall 2013 survey by TPWD, no hydrilla was noted on Lake Austin. Unfortunately, no other aquatic vegetation has been able to re-establish yet and the vegetative cover on Lake Austin has been low. This has led to a decline in the vegetative sub-index in Lake Austin over time (Figure 25).

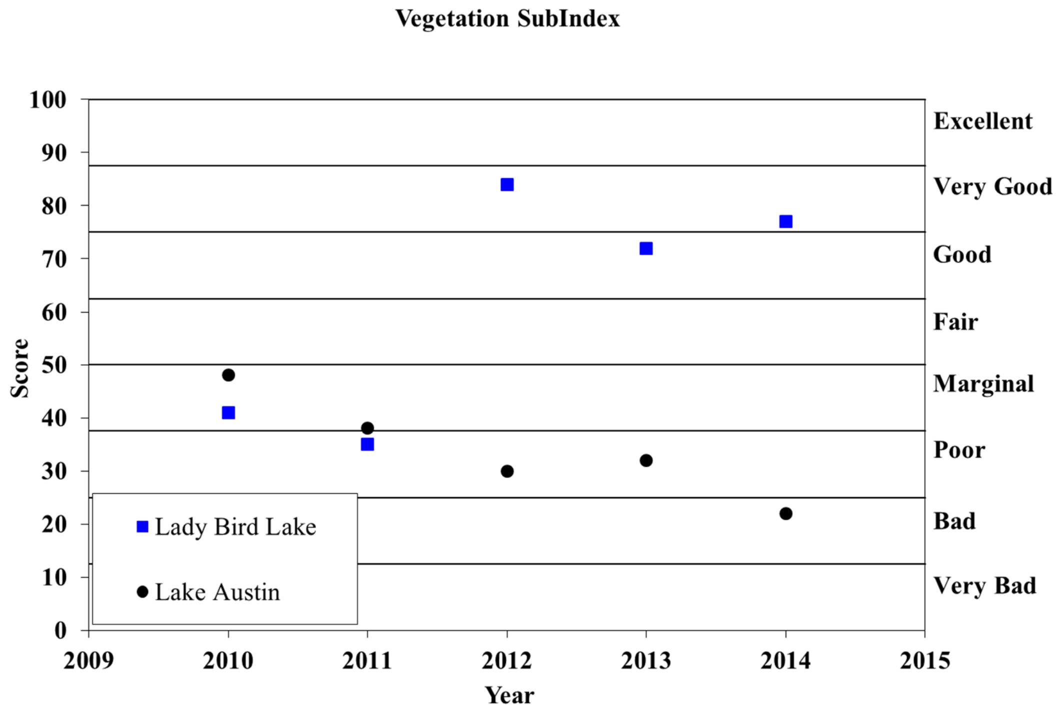


Figure 25: Vegetation sub-index of the Austin Lake Index from 2010 to 2014.

Benthic macroinvertebrate communities respond to both short and long-term environmental stresses in water, sediment, and habitat quality (EPA 2011). The community structure data collected from lake sites was transformed into qualitative metrics that describe aspects of the community (Barbour et. al. 1995), then an aquatic life sub-index was calculated based on appropriate community metrics within each lake (Richter 2011). Community metrics included in the sub-index are listed in Table 21.

Table 23: Metrics used based on multivariate analysis of the data.

Metrics Used	
# of EPT Taxa	Percent EPT
# of Taxa	Percent Dominance (Top 3)*
Percent as Tolerant Organisms*	Percent as Chironomidae*
Hilsenhoff Biotic Index*	

*indicates metrics in which high scores represent poor community health (reverse scale)

Taxa richness is representative of the diversity of the community. Increased diversity suggests that the habitat and food supply present in the system can support many different species. Two separate richness categories are used in the aquatic life sub-index, the number of taxa and the number of EPT taxa. The number of taxa has always been higher in Lady Bird Lake in comparison to Lake Austin which has usually had small communities. The number of EPT taxa is the number of insects in the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies), which are typically thought of as being sensitive to environmental stress. Again, Lady Bird Lake typically has a higher number of EPT taxa when compared to Lake Austin with Lake Austin's smaller overall community. In 2014, the number of EPT taxa increased slightly within Lake Austin.

Relative abundance is the relative contribution of individuals to the total fauna within a community and is more informative than absolute abundance for populations without knowledge of the interaction among taxa (Barbour et al. 1995). Relative abundance metrics used to calculate the aquatic life sub-index include the percent of EPT and the percent as Chironomidae. While EPT taxa are generally considered to be sensitive taxa, Chironomidae (midges) are generally considered to be tolerant and as perturbations of the environment occur the percent of Chironomidae in the community should increase. Most sites within Lady Bird Lake and Lake Austin have low to moderate percentages of EPT and percentages of Chironomidae. However, in 2013-14 sites within Lake Austin seemed to have a slightly higher percentage of Chironomidae in the community and communities in both lakes seemed to have a lower percentage of EPT in 2014.

Tolerance measures are supposed to be representative of the sensitivity to perturbation of the community (Barbour et al. 1995). Non-specific measures, do not target any type of stressor, used to calculate the aquatic life sub-index include percent dominance and the percent of tolerant organisms. The Hilsenhoff Biotic Index (HBI) is also used to calculate the sub-index and was originally designed to target organic pollution as a stressor to the community (Hilsenhoff 1987, 1988). In each lake, the communities in one site seem to be dominated by the top three most abundant taxa, while other sites have low to moderate dominance of the top three most abundant taxa. Both lakes have always retained a low percentage of tolerant organisms; however, the HBI component has typically been low within each lake indicating evidence of organic pollution. In 2014, the HBI component score improved within Lake Austin while the component in Lady Bird Lake was lower than in previous years. It is likely the change in the HBI component that caused the aquatic life sub-index to be much lower in 2014 for Lady Bird Lake (Figure 26).

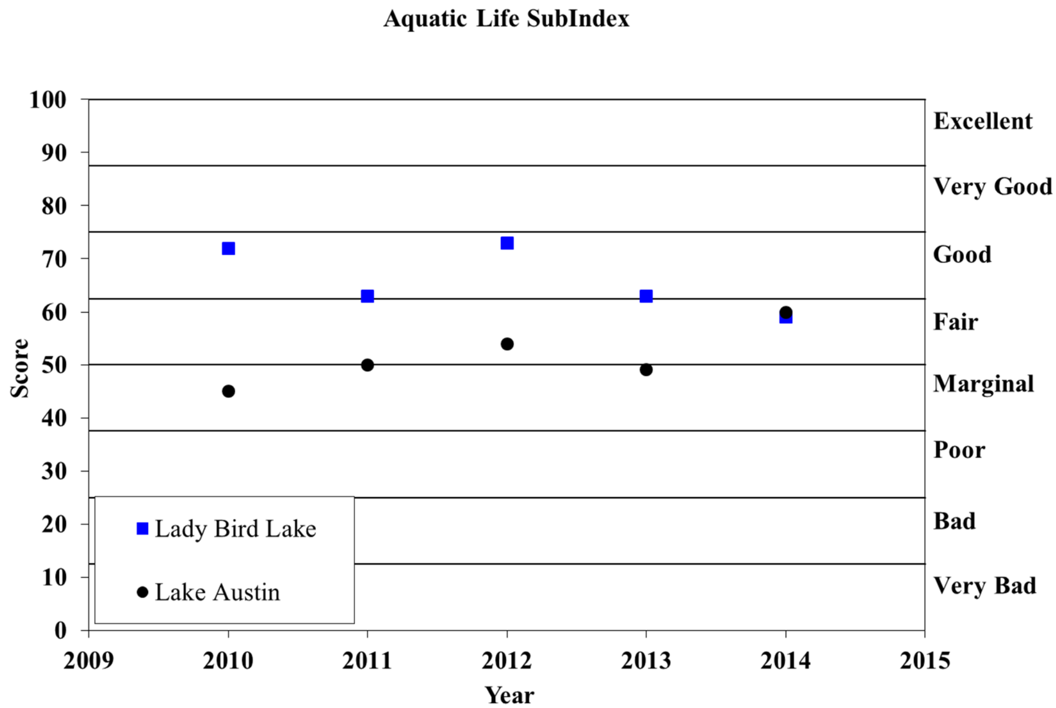


Figure 26: Aquatic life sub-index of the Austin Lake Index from 2010 to 2014.

Protection of the habitat surrounding or within any lake, including the lake substrate, shoreline characteristics, and riparian zone characteristics, is vital to maintaining the environmental integrity of the lake. Changes in aquatic and riparian habitat can lead to changes in the biological communities (i.e. fish and benthic macroinvertebrates), vegetation, and trophic status of the lake through multiple avenues including erosion and nutrient loading. The habitat sub-index is based on the substrate conditions, aquatic cover present along shorelines, shoreline characteristics, and riparian vegetation present along each lake (Richter 2011).

The substrate in both lakes has typically been noted as being dominated by sand or silt. Sand and silt were designated as undesirable substrates because they represent sedimentation of smaller particle sizes which is known to degrade habitats by lowering interstitial dissolved oxygen and reducing benthic production (Chapman 1988). Thus, the substrate component of the habitat sub-index has typically been moderately low in both lakes.

Available cover along the shoreline influences aquatic organisms by providing shelter and an influx of organic matter (Angermeier and Karr 1984, Benke et. al. 1985). Lady Bird Lake has been typically described with moderate amounts of submerged and emergent macrophytes, woody debris, and overhanging vegetation, but in 2014 there was a decrease in the amount of emergent macrophytes and overhanging vegetation observed along the lake. Lake Austin has typically been described with moderate amounts of woody debris and overhanging vegetation with sparse coverage of human structures along the shore. In 2014, there was a decrease in the amount of observed macrophyte coverage and an increase in the amount of human structures observed along the shore. The change in the aquatic cover component in 2014 led to a decrease in the habitat sub-index score in 2014 (Figure 27).

Riparian cover along Lady Bird Lake is moderate in the canopy, understory, and groundcover layer of vegetation with few observations of invasive species. Vegetation cover in the riparian zone along Lake Austin is moderate in the canopy but sparse in many locations for understory and groundcover vegetation. Very few invasive species are observed in the riparian zone along Lake Austin, in fact there were no observations of invasive species in the 2014 survey. The riparian zone component of the habitat sub-index is typically higher in Lady Bird Lake due to the increased vegetative cover in the riparian zone along the lake.

The shoreline substrate along each lake is dominated by fine sediments, similar to the substrate within each lake, with some terrestrial vegetation and cobble. In addition, both lakes are impacted by a moderate amount of human structures or disturbances along the shoreline. One key difference to the shoreline characteristics between the lakes is the shoreline slopes. Lady Bird Lake has gentle slopes which typically relates to less erosion potential. Lake Austin has more steeper slopes with more erosion potential coupled with vertical bulkheads in some locations which leads to more environmental degradation. Thus the shoreline component of the habitat sub-index is always lower in Lake Austin than in Lady Bird Lake.

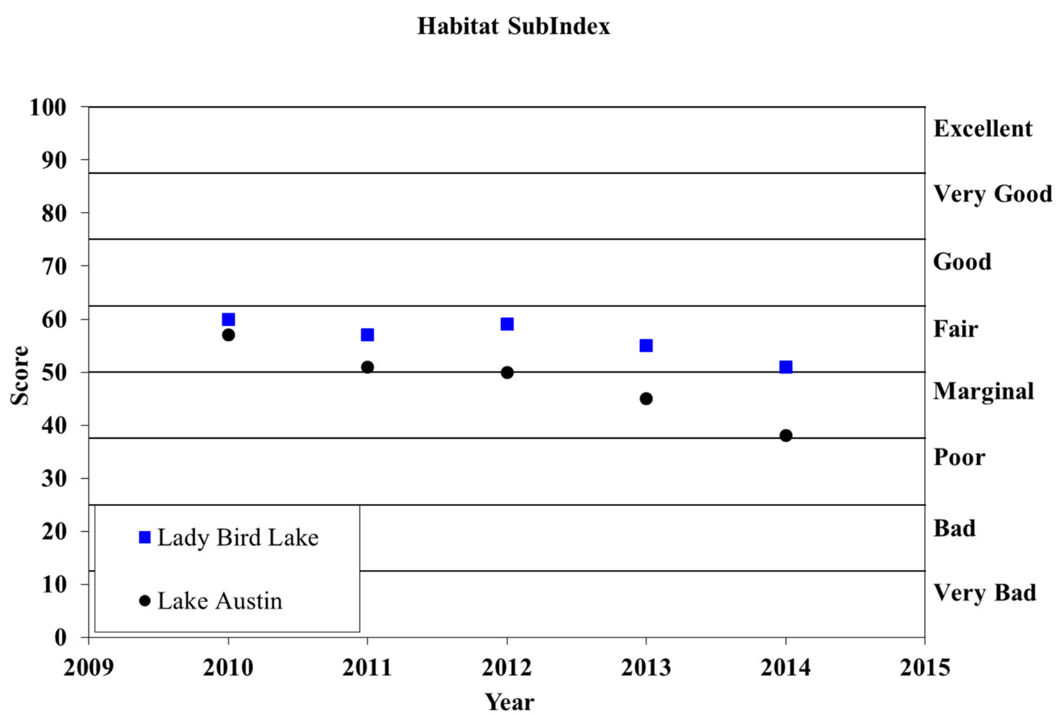


Figure 27: Habitat sub-index of the Austin Lake Index from 2010 to 2014.

Eutrophication has been defined as the movement of a water body's trophic status in the direction of more plant biomass (Carlson and Simpson 1996). This can include increased algal biomass, macrophyte biomass, and nuisance algae blooms which lead to a decreased aesthetic appeal, decreased number of desirable game fish, loss of accessibility, and increased cost of drinking water treatment (EPA 2011). The eutrophication component of the ALI is based on the concentration of chlorophyll a and the phytoplankton community composition, specifically the percent of the community represented by cyanobacteria (blue-green algae), green algae, diatoms, and chrysophytes (Richter 2011). As noted earlier, as the chlorophyll a concentration increases the trophic status of the lake would pass from oligotrophic to mesotrophic to eutrophic and the eutrophication sub-index would decrease. Similarly, the percent of cyanobacteria and green algae in the community should increase as eutrophication occurs while the percent of diatoms and chrysophytes should decrease. The ALI is calculated so that high percent composition of cyanobacteria or green algae translates to a lower score while high percent composition of diatoms or chrysophytes translates to a higher score.

As noted earlier, the cyanobacteria blooms are increasing in frequency and intensity; however, the eutrophication sub-index of the ALI has not responded to this increase. The eutrophication sub-index for Lake Austin has been stable since 2010, while the eutrophication sub-index for Lady Bird Lake has displayed a minor decrease (Figure 28). It is possible that the simultaneous use of the cyanobacteria and green algae communities with the diatom and chrysophyte communities has led to an undesirable averaging effect on the eutrophication sub-index of the ALI. Analyses below examine the eutrophication sub-index to determine if there is a more appropriate method for calculating this sub-index.

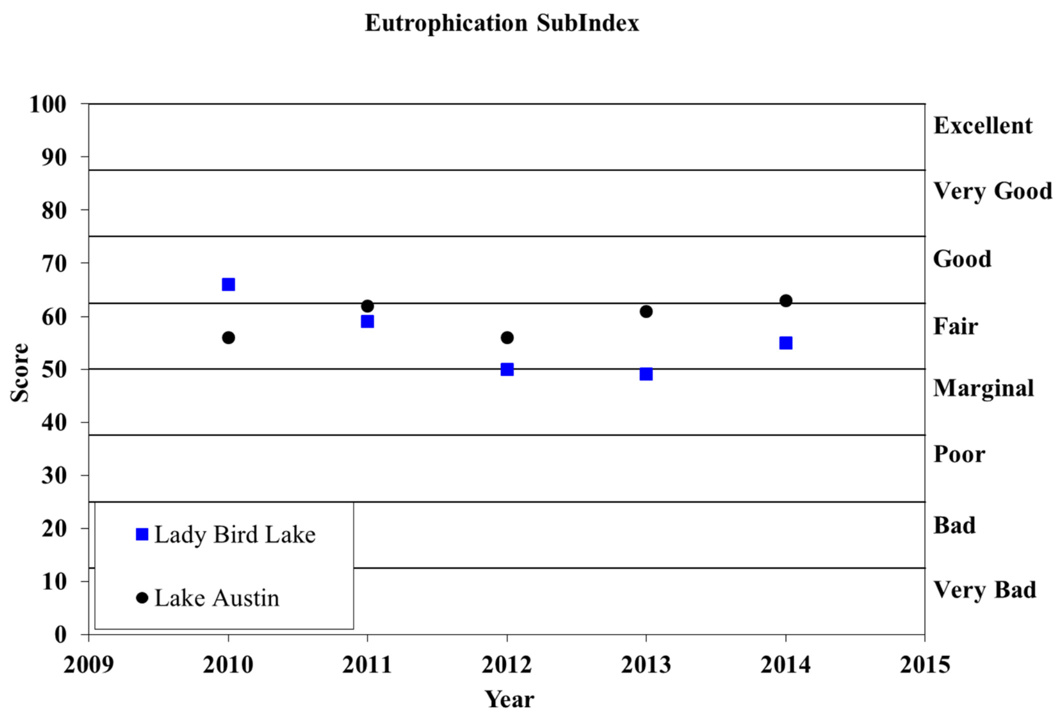


Figure 28: Eutrophication sub-index of the Austin Lake Index from 2010 to 2014.

Phytoplankton data for the Austin Lake Index is collected in April, June, August, and October. Logistic regression was used to determine if these months had the highest probability of algal blooms. The logistic model at the Davis WTP predicted the occurrence of a total phytoplankton bloom fairly well with a 0.798 *c-statistic*. While this is not an ideal fit, the model should be good enough to use for examining the months with the highest probability of a total phytoplankton bloom. The months with the highest predicted probability of a total phytoplankton each year were July, August, September, and October; with mean probabilities of 0.22, 0.31, 0.28, and 0.23 respectively (Appendix F). The 2013 probabilities for each of the above months were 0.67, 0.78, 0.74, and 0.69 respectively. In some years, April and June also had higher probabilities of a total phytoplankton bloom.

The logistic model predicted the occurrence of a blue-green algae bloom very well with a 0.960 *c-statistic*. The months with the highest predicted probability of a blue-green algae bloom were August, September, and October; with mean probabilities of 0.21, 0.30, and 0.39 respectively (Appendix G). The 2013 probabilities for each of the above months were 0.68, 0.79, and 0.87 respectively. In certain years, especially more recent years, July and November also had high probability of a blue-green algae bloom. The logistic model also predicted the occurrence of a diatom bloom very well with a 0.963 c-statistic. The months with the highest predicted probability of a diatom bloom were September and October; with mean probabilities of 0.35 and 0.40 respectively (Appendix H). The 2013 probability in September was 0.86 while the probability was 0.89 for October. In some years August also had a higher predicted probability and in other years July and November also showed higher predicted probabilities for a diatom bloom.

Water quality data is currently collected in April, June, August, and October for the Austin Lake Index (ALI). Of the months sampled, April and June have typically had the lowest probabilities for phytoplankton, blue-green, or diatom algae blooms. August and October have had higher probabilities for blooms. One goal of the ALI is to be able to compare eutrophication (phytoplankton data) from one year to the next. In order to do this, phytoplankton should be collected in roughly the same state from year to year (bloom vs non-bloom). It is most likely that phytoplankton collected in August and October would be in a bloom period, but the state of phytoplankton in April and June is more variable. A more optimized sampling schedule may be required in order to properly track eutrophication changes in the lakes.

The current Eutrophication Index in the ALI is calculated using the total chlorophyll *a* in the water column along with blue-green algae, green algae, diatoms, and chrysophyte counts. Data collected at the Davis WTP lends itself to analyzing the blue-green/green and diatom counts. Currently one grab sample from April, June, August, and October is collected to calculate the ALI. If data was collected every day in the above months, the ALI score would fully represent the phytoplankton conditions present in the lake during April, June, August, and October. Sampling each day is cost and time prohibited, but increasing the frequency of sampling within April, June, August, and October might allow the COA to capture phytoplankton communities in more than one state (bloom vs non-bloom) if they existed and produce more accurate ALI scores. The Blue-green Index, Diatom Index, and Eutrophication Index were computed for each day using the phytoplankton data from the Davis WTP in 2012. The year 2012 was chosen because it was a recent year with more variability in the predicted bloom probability than 2013. A data scheme was set up where days were randomly combined into 1, 2, 3, 4, and 5 samples a month for April, June, August, and October. The range of the Blue-green, Diatom, and Eutrophication Index decreased with increasing sample frequency in each month (Table 24). The Blue-green Index range decreased from 10 at one sample every month to about three at five samples a month while the other indices decreased from a range of about 10 at one sample every month to about five at five samples a month.

Table 24: Blue-green, Diatom, and Eutrophication Indices for the Davis WTP in 2012 when 1, 2, 3, 4, or 5 samples were collected in April, June, August, and October.

Index	Number of Samples Each Month	Mean Index Score	Standard Deviation	Range of Scores
Blue-green Index	1	85.6	2.4	79.6 – 89.9
	2	85.7	1.5	82.1 – 88.4
	3	85.7	0.93	83.6 – 87.8
	4	85.7	0.89	84.0 – 87.1
	5	85.7	0.82	84.3 – 87.5
Diatom Index	1	14.8	4.2	9.2 – 22.6
	2	14.8	2.3	10.6 – 19.2
	3	14.7	1.7	11.7 – 18.5
	4	14.7	1.9	12.2 – 18.2
	5	14.7	1.5	12.5 – 17.7
Eutrophication Index	1	50.2	3.0	46.7 – 56.1
	2	50.2	1.6	47.6 – 53.3
	3	50.2	0.93	48.3 – 52.0
	4	50.2	1.1	48.7 – 52.5
	5	50.2	0.88	48.7 – 52.0

Another way for the COA to be more confident that the ALI score has accurately represented phytoplankton growth over a given year through the Eutrophication score would be to add sampling in more months. In order to determine how additional sampling in months other than April, June, August, or October would affect the ALI scores, one hundred random samples were generated from data collected at the Davis WTP for the months of April, May, June, July, August, September, October, and November. These months were chosen because they have higher predicted probabilities in some years including 2012 which was the year used for analysis. A Blue-green Index, Diatom Index, and Eutrophication Index were calculated for each random sample. The base score for each random sample was calculated as the average score from April, June, August, and October. Each month was averaged in to the base score until all combinations of additional monthly sampling were analyzed (Table 25). Adding months to the sampling scheme did not dramatically alter the mean or the range of the Blue-green algae or Diatom Index. If new months were added to the sampling protocol an investigation of how the new score could relate to the old score may be necessary, but this analysis showed that adding sampling in May, July, September, or November would probably not alter the index scoring greatly.

An index was calculated for every day in April, June, August, and October for each year from 1992 through 2013 based on a combination of blue-green algae and diatoms and another index was calculated which was strictly based on blue-green algae. Average index scores were calculated each year using indices calculated only in April, June, August, and October. The index calculated using the blue-green algae and counts and the diatom counts were much lower than the index that only used the blue-green counts. In most instances the diatom counts were a low percentage of the phytoplankton community and lowered the score substantially.

Table 25: Average Blue-green sub-index, Diatom sub-index, and Eutrophication Index with standard deviation and ranges for COA current sampling and additional monthly sampling.

Model	BG Index			Diatom Index			Eutrophication Index		
	mean	sd	range	mean	sd	range	mean	Sd	range
base	85.9	2	81.1-90.5	14.9	4	8.6-24.7	50.4	2.5	46.2-57.5
may	87.3	1.6	83.9-90.8	15.5	3.5	7.8-25.2	51.4	2.1	46.9-57.9
july	83.6	1.9	79.3-88.5	16.7	3.4	11.1-24.6	50.2	2.1	46.2-55.6
sept	84.9	1.8	80.3-89.3	13.5	3.4	8.1-21.6	49.2	2.2	45.2-54.8
nov	87.3	1.6	83.9-91.4	13.6	3.4	7.7-21.6	50.4	2	47.0-56.0
may july	85.2	1.6	81.6-89.7	16.9	3	9.9-25.0	51.1	1.9	47.5-56.3
may sept	86.3	1.6	82.6-90.5	14.2	3.1	7.7-22.2	50.3	1.9	46.2-55.6
may nov	88.2	1.4	85.5-91.5	14.3	3	8.1-21.9	51.3	1.8	47.4-56.6
july sept	83.2	1.8	78.8-88.5	15.3	2.9	10.2-22.5	49.2	1.9	45.7-54.3
july nov	85.1	1.5	81.8-88.5	15.3	2.9	10.1-22.5	50.2	1.8	47.2-54.6
sept nov	86.3	1.5	82.4-89.5	12.6	3	7.4-21.6	49.4	1.9	46.3-54.1
may july sept	84.6	1.6	81.2-89.7	15.6	2.7	9.2-22.5	50.1	1.7	46.8-54.5
may july nov	86.3	1.4	83.3-89.7	15.7	2.7	9.3-22.2	51	1.6	47.8-55.4
may sept nov	87.2	1.3	84.1-90.5	13.4	2.8	7.9-20.3	50.3	1.7	46.7-54.8
july sept nov	84.6	1.5	80.9-88.5	14.2	2.7	9.5-22.5	49.4	1.7	46.6-54.3
may july sept nov	85.6	1.4	82.7-89.7	14.7	2.5	8.8-20.9	50.2	1.5	47.1-53.9

The two indices track fairly well with each other with a few notable exceptions including 1992, 2012, and 2013 (Table 26, Figure 29). In these years the samples collected had higher diatom percentages in the communities which increased the score of the index based on both types of algae communities. Meanwhile, Lake Austin experienced one of the worst blue-green algae blooms in the past decade in 2013 which would not be obvious in the index based on both algae types. The diatom community was originally added to the ALI Eutrophication Index because it was thought that an increase in the percentage of diatom counts would indicate increased water quality, but it may be that the inclusion of both blue-green algae and diatoms has convoluted the scoring index. The Eutrophication Index may need to be revisited in the ALI to better track changes in water quality.

Table 26: Average index with 95% confidence intervals by year based on blue-green algae counts with diatom counts or based on blue-green algae counts alone.

Year	**Blue-green/Diatom Index**			**Blue-green Index**		
	Index	**95% CI**		**Index**	**95% CI**	
1992	52.9	45.6	60.1	90.1	85.9	94.4
1993	51.5	50.5	52.5	95.4	94.3	96.4
1994	51.0	48.4	53.5	95.3	94.2	96.4
1995	52.7	49.9	55.4	96.2	95.0	97.5
1996	48.8	47.3	50.2	93.0	91.0	95.1
1997	51.6	50.2	53.1	93.1	90.7	95.5
1998	50.0	49.0	51.0	95.3	93.7	97.0
1999	51.9	50.6	53.2	96.4	95.4	97.4
2000	50.2	49.1	51.2	95.4	94.1	96.8
2001	51.7	50.5	52.8	96.1	95.0	97.2
2002	49.0	46.9	51.2	91.3	87.5	95.1
2003	51.1	48.2	54.0	94.5	89.7	99.2
2004	48.3	46.6	50.1	91.8	88.3	95.4
2005	54.4	52.1	56.7	96.0	94.8	97.2
2006	48.3	46.7	49.8	90.4	88.1	92.7
2007	47.3	44.8	49.8	85.4	81.3	89.4
2008	54.6	52.8	56.4	95.2	94.1	96.2
2009	44.5	42.8	46.3	84.5	81.5	87.4
2010	53.9	52.3	55.5	94.2	92.9	95.4
2011	50.2	48.6	51.8	90.1	88.4	91.8
2012	50.2	48.4	52.1	85.8	84.1	87.4
2013	51.5	48.2	54.8	81.0	77.6	84.3

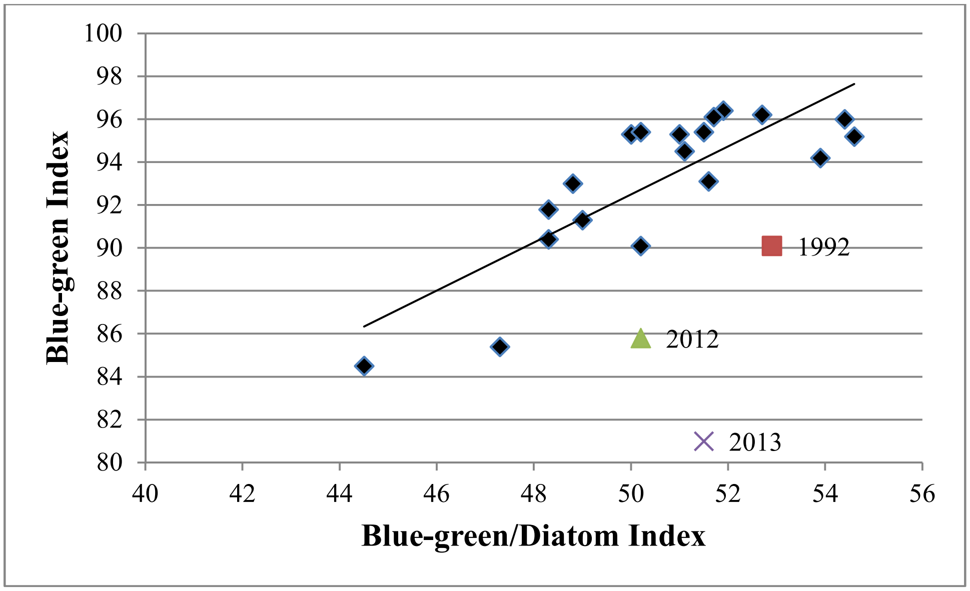


Figure 29: Yearly average index scores calculated using blue-green and diatom counts (x-axis) and only blue-green counts (y-axis). The years 1992, 2012, and 2013 appear as outliers.

Conclusions/Recommendations

- Estimates of the mean concentration of various water quality parameters in Lady Bird Lake and Lake Austin have been calculated from over a decade of data collection. These estimates of mean concentrations give an accurate indication of the water quality of the lake and can be used as *a priori* values in future lake studies which may include looking at trends or impacts on the system.

- From these estimates, it appears that Lady Bird Lake and Lake Austin all have more degraded levels of nitrate, dissolved oxygen, temperature, and *E. coli* in downstream locations versus upstream locations. Furthermore, TCEQ water quality criteria verify that these water quality parameters are at levels that may be of concern or may be impaired. Locations of concern in Lady Bird Lake include areas downstream of the 1st Street Bridge. Similarly, for Lake Austin, locations of concern include the most downstream areas (that is, areas downstream of Bull Creek). Additionally, in Lake Austin, the area around Selma Hughes Park has more degraded concentrations of dissolved oxygen than upstream or downstream locations.

- Blue-green algal bloom intensity has increased over time between 1992 and 2013 at both the Davis and Ullrich WTP on Lake Austin. More specifically, high blue-green algae counts increased in intensity from 2004 to 2009. A brief decrease in blue-green algae numbers occurred during 2010 and 2011, but was followed by increasing counts in 2012 and 2013.

- Four of the six years with the highest probability of an algal bloom took place from 2009 to 2013. Thus higher blue-green algal counts are becoming more frequent on Lake Austin.

- Increases in phytoplankton or blue-green counts may be attributed to relationships with phosphate concentrations and flow. Counts increase as the flow decreases in the reservoir. If management of the reservoirs continues to reduce the amount of flow through the system by not releasing water downstream, conditions will be better suited for more algal growth. Better resolution of phosphorus data is needed to fully understand the relationship between the phytoplankton and phosphorus concentrations. Multiple imputation was used to form a theoretical model, but phosphate and other nutrients should be collected on more frequent intervals to validate the model formed.

- Further analysis of the phytoplankton time series data is warranted. Generalized dynamic linear models are a form of models that have been shown to be able to model time series data of irregular time intervals with non-linear, non-Gaussian data. These can be used to separate the noise in the data from trends and it is recommended that further efforts be made in the future to model the phytoplankton counts at the Davis and Ullrich WTP together in a generalized dynamic linear model using Bayesian inference so that the data at each site can "borrow strength" from the other data set.

- ALI data is currently collected in April, June, August, and October. Two of these months fall within the highest probable months for an algal bloom (August and October), especially a blue-green bloom.

- Adding more samples to the ALI data collected in April, June, August, and October could increase the accuracy of the Eutrophication Index substantially.

- Adding more samples to the ALI data to be collected in other months may alter the scoring by analyzing different phytoplankton communities; however, adding sampling in May, July, September, or November may not alter scores greatly.

- Taking out the Diatom sub-index from the Eutrophication Index would allow for better tracking over time of blue-green algae trends. The Eutrophication Index should be revisited by staff to decide if tracking only blue-green algae trends would better represent water quality on the lakes or if the current community composition methodology is sufficient.

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